MANAGEMENT AND CREATION OF OPEN MOSAIC HABITAT FOR INVERTEBRATE CONSERVATION

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Declaration

I certify that this thesis represents my own work, unless otherwise identified by references. It has not been submitted to another institution, or accepted for any degree, other than the Doctor of Philosophy being studied at the University of East London.

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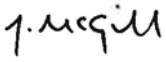
Signature: 

Abstract

Some brownfield sites can support comparable invertebrate diversity to semi-natural early successional habitats. This was recognised in the designation of Open Mosaic Habitat on Previously Developed Land as a UK conservation priority. This project developed in response to the need for information to assist with management of brownfields protected for nature conservation, and from the lack of evidence about effectiveness of brownfield mosaic habitat creation. The study included: management interventions at Canvey Wick in Essex, a brownfield nature reserve; mosaic habitat creation by substrate addition in unmanaged grassland; and placement of artificial aculeate nests on green roofs in London.

Scrub clearance at Canvey Wick produced distinct communities of stenotopic invertebrates immediately after management, and one year later. Trait-based analysis showed that staphylinids and spiders were larger at undisturbed plots, dispersal traits for carabids and spiders were linked to disturbance, and trophic shifts for all three groups. In species-poor grassland at Canvey Wick, stenotopic beetles particularly benefitted from scraping, whereas stenotopic and generalist spiders were more abundant after excavator disturbance, and at unmanaged plots. Carabids and staphylinids were larger at undisturbed plots, although treatment responses in leg and eye morphology were not shared.

Substrate addition in unmanaged grassland on clayish soils in south Essex benefitted nesting aculeate Hymenoptera, particularly in the first year. Stenotopic early successional beetles and spiders were more abundant in the second year, although species composition was similar to the grassland. Cavity nesting aculeate Hymenoptera were reared from biodiverse and unvegetated roofs in London, although most sites were not colonised, and substrate nests were unsuccessful.

Results were examined in relation to broader literature concerning invertebrate conservation in early successional habitats. Implications for land managers were discussed, including species turnover, habitat structure, and potential benefit to ecological networks from habitat restoration, management and creation in appropriate locations.

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Abbreviations

CBD: Convention on Biological Diversity

CCA: Canonical correspondence analysis

DCA: Detrended correspondence analysis

Defra: Department for Environment, Food and Rural Affairs

DoE: Department of Environment

F_{Dis}: Functional dispersion

F_{Div}: Functional divergence

F_{Eve}: Functional evenness

GLM: Generalized linear model

GLMM: Generalized linear mixed model

NMDS: Non-metric multidimensional scaling

OMH: Open Mosaic Habitat on Previously Developed Land

PFA: Pulverised fuel ash

PDL: Previously developed land

PCoA: Principal coordinates analysis

RDA: Redundancy analysis

RSPB: Royal Society for the Protection of Birds

SSSI: Site of Special Scientific Interest

UK: United Kingdom

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Chapter 1 – Introduction

1.1 Conservation background

The modification and fragmentation of terrestrial habitats is a key cause of global biodiversity loss and degradation of ecosystem services (Fischer and Lindenmayer, 2007). The Convention on Biological Diversity (CBD) opened at the 1992 United Nations 'Earth Summit' was an international political response to these changes. This is a legally binding agreement with 194 current signatories, and includes the conservation of biodiversity as a main goal. To implement the Convention, the United Kingdom (UK) published a Biodiversity Action Plan (Department of Environment [DoE], 1994). This described national biological resources, as well as strategies for conservation of the most threatened habitats and species. The current CBD Aichi targets include integration of biodiversity in decision-making, and safeguards for ecosystems and ecological processes, which are also adequately represented within protected areas (Secretariat of the Convention on Biological Diversity, 2014). Translated into current national policy, there is an emphasis on ecological networks (Lawton *et al.*, 2010), and the goal of 'no net loss' for priority habitats (Department for Environment, Food and Rural Affairs [Defra], 2011).

General trends in most European regions during the twentieth century have reduced the conservation importance associated with early successional stages (Morris *et al.*, 1994). These include declines in traditional agricultural management (Prévosto *et al.*, 2011) and the effect of myxomatosis on rabbit populations (Millett and Edmondson, 2013). In England, the area of heathland decreased by 84% after 1800 (English Nature, 2002) and 47% of semi-improved grasslands were lost between 1960 and 2013 (Ridding *et al.*, 2015). Sedimentary structures along river channels and floodplains have been affected by engineering works, including channel deepening, realignment, embankment and impoundment (O'Callaghan *et al.*, 2013). Coast protection and stabilisation have impacted early successional invertebrates at some soft rock cliffs and slopes (Howe, 2015). Agricultural improvement has also reduced the capacity of farmland to support biodiversity (Wilson *et al.*, 1999; Benton *et al.*, 2003). Overall, wildlife-rich sites are isolated in a landscape which lacks the habitat diversity or resources to support sustainable populations of many species (Lawton et al., 2010).

Against this background, artificial analogues for semi-natural habitats have developed in some urban and industrial areas (Eversham *et al.*, 1996). This is possible where species depend on functional habitat structures, which are shared between superficially different ecosystems. The butterfly *Hamearis lucina* has been associated with open deciduous woodland, scrubby calcareous grassland, and chalk quarries (Turner *et al.*, 2009). These situations all provide larger foodplants (*Primula*) preferred by the larvae, and shelter for adults, which fly in spring when daytime temperatures may be relatively low. Similarly, riparian ground beetles may also be found at gravel pit margins, where requirements for microclimate and substrate are matched (Gerken *et al.*, 1991). Semi-natural habitat loss and fragmentation can result in artificial analogues becoming important for conservation at a national level (Bogusch *et al.*, 2016). At least 15% of invertebrates with UK conservation status were estimated to occur in artificial ecosystems, although with uneven geographical and taxonomic distribution (Gibson, 1998). Grassland, heathland and coastal habitats were particularly well represented, compared with uplands, wetlands and ancient woodland (Gibson, 1998).

Some of the most important artificial sites for biodiversity conservation are brownfield land. This term was popularised in the early 1990s in the United States, describing land with real or perceived contamination that was an obstacle to redevelopment (Adams *et al.*, 2010). In the UK, brownfield is more widely defined as previously developed land that is unused, or potentially available for redevelopment (Office of the Deputy Prime Minister, 2005). This includes vacant and derelict land, infill sites, and land occupied by redundant or unused buildings (The Scottish Government, 2008). Farmland, recreational areas, and domestic gardens are excluded (Bodsworth *et al.*, 2005). In England, there were large regional differences in availability of previously developed land in 2012, from 1,240 ha in London, to 7,200 ha in the North-west (Sinnett *et al.*, 2014). This is the last year with data, as Government has not commissioned subsequent surveys. Previously developed land is also a dynamic resource, determined by the balance between use and replenishment of sites (Strauss and Biedermann, 2006). Brownfield re-use was 30% more likely in London in 2006, compared with North-west England (Longo and Campbell, 2017).

Not all brownfields are valuable for biodiversity, including some buildings, areas of hard-standing, and car parks (London Wildlife Trust, 2002). A subset of wildlife-rich sites was characterised in the UK Biodiversity Action Plan, as Open Mosaic Habitat on Previously Developed Land (OMH) (Maddock, 2011). In the post-2010 UK biodiversity framework, priority habitats are listed under Section 41 of the Natural Environment and Rural Communities Act, and included in the duty of every public authority under Section 40 to have regard for the conservation of biodiversity (Joint Nature Conservation Committee and Defra, 2012). Sites qualifying as OMH have a history of human disturbance, or evidence that soil has been removed or severely modified, in an area of at least 0.25 ha (Maddock, 2011). Extraneous substrates may also have been added. This produces a habitat mosaic of bare ground in association with early successional plant communities. These include annuals, bryophytes, lichens, open grassland, flower rich grassland, and ruderal, inundation or heathland vegetation. Irregular terrain, pools, scattered scrub, and rough grassland may be present, and add to the conservation value of a site. Habitat mosaics are particularly valuable for invertebrates, and also plants, lichens, birds, reptiles, and amphibians (Macadam *et al.*, 2013).

Complete field data on the extent of Open Mosaic Habitat is lacking (Natural England, 2008). The most recent estimate in England, collated from the National Land Use database, British Pits database, and Historic Landfill data, is of 7% of brownfield as OMH, or 48,758 ha (Davenport, 2014). This was derived from the percentage of a subset of points checked, that resulted in some OMH being mapped into a national inventory (Lush *et al.*, 2014). In Scotland, 67 of 382 sites on the Vacant and Derelict Land Register in 2009 were identified as OMH from aerial photography (Macadam *et al.*, 2013). Wales only has a partial OMH inventory and no estimates are available for Northern Ireland. In England, the area of OMH compares with 56,819 ha lowland heathland, and 65,567 ha lowland calcareous grassland (Defra, 2011). Both habitats are highly valued for nature conservation, with 41,015 ha and 45,847 ha within Sites of Special Scientific Interest (SSSI), respectively (Defra, 2011). OMH is currently classified by existing semi-natural habitat categories, so it is not possible to quantify the area within SSSIs, but this is probably much lower (Lush *et al.*, 2014).

1.2 Brownfield ecology

1.2.1 Substrate and topographical structures

Conservation management at brownfields requires a detailed understanding of the development of faunal diversity. A potential model of anthropogenic and ecological processes links the effects of disturbance to topographic structure and vegetation succession (Figure 1.1). Substrate provides physical habitat for invertebrate colonisation, at brownfields frequently as a by-product of human economic activity. Some industrial processes generate large quantities of solid waste, which were historically left to revegetate as mounds. Pulverised fuel ash (PFA) is a fine-grained ash produced in coal combustion (Watt and Thorne, 1965). LeBlanc waste is an insoluble mixture of calcium sulphide and hydroxide generated during production of sodium carbonate from salt, although this process was defunct by the early twentieth century (Ash *et al.*, 1994). Other by-products include lime waste, blast furnace slag, and mine spoil (Shaw, 1998). Brick rubble sites have been created after demolition of housing, or factory complexes following industrial decline (Gilbert, 1989). Extraction is a third source of exposure for nutrient-poor substrates, including minerals (Prach *et al.*, 2011) and peat (Hodge, 1989).

Substrates cover a range of environmental values, which can differ within and between localities. There may be a mixture of chemical particles, and precipitates formed by biochemical processes (Capilla *et al.*, 2006). Higher phosphorous content at one PFA dump was explained by variation in chemistry of the source coal, compared with three other sites (Shaw, 2003). Dredging deposits may be highly varied, including sediments from the present environment, and post-glacial conditions. These grade from coarser materials to fine silt with increasing distance from the pipe outlet. At Rhode Island Sound, deposits included silts with up to 12% organic matter, estuarine silty sands with shells and pebbles, and varved sands and clays deposited in glacial outwash (Saila *et al.*, 1972). Mounding can create complex topography, including patterns of hummocks and hollows (Hall, 1957). On some substrates these support hydrological structures, including ephemeral and permanent water bodies (Doležalová *et al.*, 2012). Quarrying exposes underlying substrate, including hard rock formations, sand and gravel deposits. This is associated

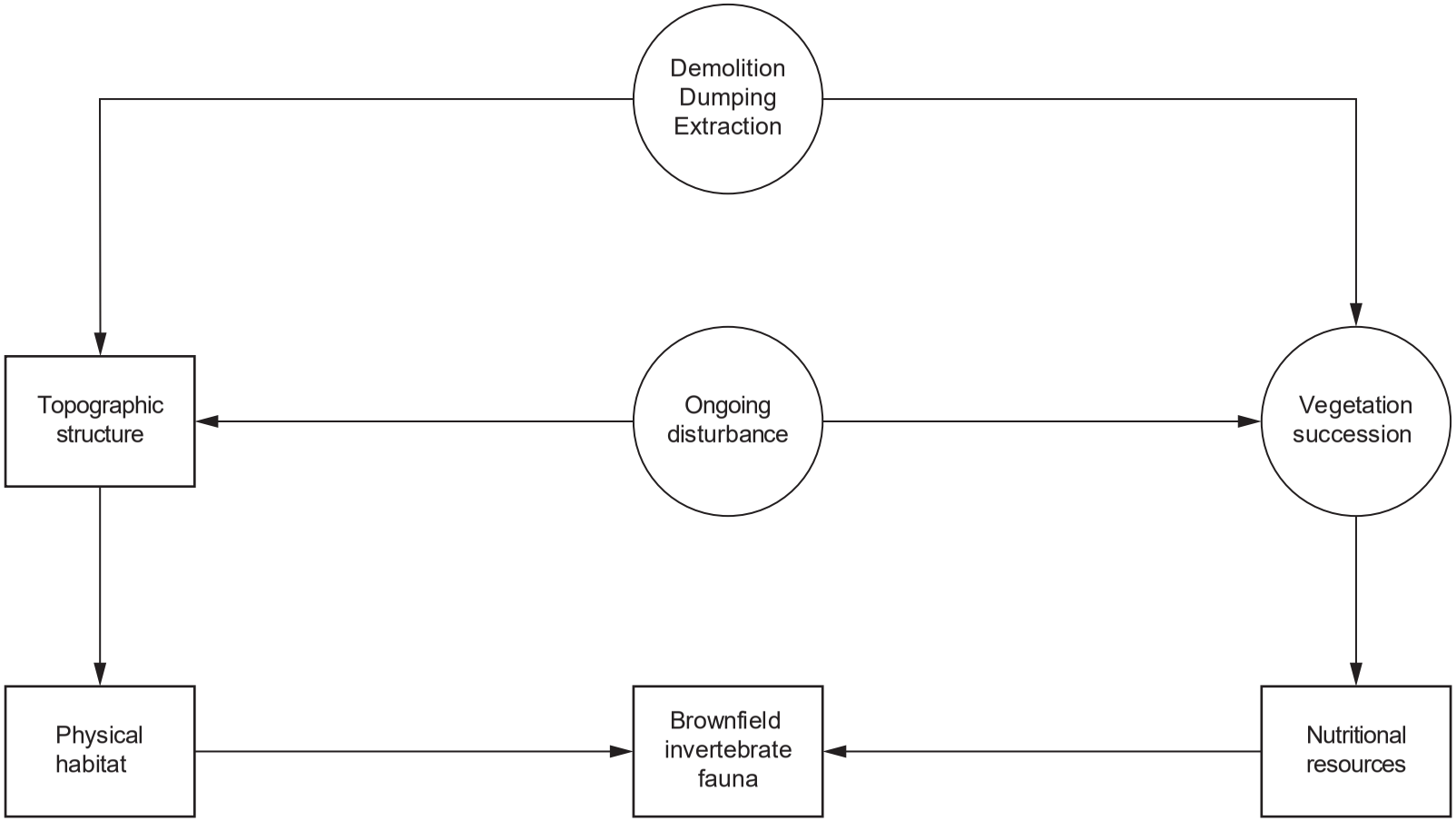

Figure 1.1 A framework for interaction of brownfield habitat structures and processes (based on Gregory *et al.*, 1991). Arrows represent predominant influences of processes (circles) on structures (rectangles). In this framework, mechanisms of brownfield site creation are linked to the development of topography and vegetation. These can be further modified by ongoing disturbance, forming dynamic physical habitat and nutritional resources for invertebrate communities. There are opportunities for research in the manipulation of the processes, to maximise benefits for invertebrate conservation.

with greater topographical modification, particularly vertical or near-vertical slopes.

1.2.2 Vegetation

Substrate also influences brownfield invertebrate communities indirectly through the development of vegetation. The chemical composition of some materials may inhibit plant establishment. Substrate pH affects plant growth mainly through effects on the solubility of chemicals, including nutrients and toxic metals (Wong, 2003). PFA is initially highly saline (conductivities up to 50 mS/cm) due to dissolved sodium, potassium, sulfate, chloride, and borate, in addition to high boron content and negligible nitrogen (Shaw, 2009a). Elevated concentrations of metals in mine tailings can include arsenic, cadmium, copper, manganese, lead and zinc (1–50 g/kg) (Boulet and Larocque, 1998). LeBlanc waste is characterised by extreme phosphate binding, resulting in sites with less than 50% vegetation cover after 50 years (Bradshaw, 1983). The factors driving vegetation succession are site-specific but include the effects of rainfall weathering substrate (Shaw, 1996), accumulation of organic matter (Shaw, 1992), atmospheric deposition of nitrogen (Stevens *et al.*, 2016) and activity of nitrogen-fixing legumes (Chapin *et al.*, 1994).

Different phytosociological communities develop according to substrate, with soil chemistry and water content particularly important (Shaw, 2011). Substrates with elevated pH (PFA, LeBlanc, and Solvay process wastes) are characterised by initial growth of halophytes, then establishment of Fabaceae and *Dactylorhiza* orchids. Calcareous grassland remained stable for over 50 years on dry Solvay waste, whereas closed canopy woodland (*Betula* and *Salix*) developed on PFA after about 30 years. At derelict sandy brownfields, brick rubble was associated with higher pH and nutrient supply, and reduced water permeability (Schadek *et al.*, 2009). Plant species richness was greatest after five (± two) years, with secondary woodland established after about 40 years. The ecological trajectory is different on acidic materials, with herbs and graminoids later replaced by acidophilic grasses (e.g. *Agrostis capillaris*) and succession to scrub dominated by *Crataegus monogyna* and *Quercus robur* between 50-100 years (Hall, 1957). Plant species richness is lower at acidic sites, although notable bryophytes have been found on coal and lead spoil (Allan *et al.*, 1997).

Centripetal encroachment has been documented in experimental manipulations of vegetation succession on PFA (Shaw, 2009a). After 10 years bare ground was limited to the centre of the site where soil nitrate was lowest, although it was unclear whether this was a consequence or cause of vegetation changes. The site was also colonised from the edge by clonal species (*Hippophae rhamnoides* and *Rubus fruticosus* agg.). There are topographic effects on vegetation distribution, mediating environmental factors including exposure and moisture levels. At coal spoil heaps, elevation and aspect produced a repeated series of micro-topographically induced differences (Hall, 1957). Cover and abundance were greatest on lower south and west aspects, as well as in hollows and gullies. Bimodal or multimodal particle size distributions also encourage different patterns of plant colonisation (Sadler *et al.*, 2011). These factors have implications for the design of brownfield habitats, or a disturbance regime to maximise the persistence of early successional stages.

1.2.3 Disturbance

In addition to substrate and vegetation, brownfield ecology is shaped by disturbance. Planned disturbances are part of normal economic activity, such as repeated tipping extending a mine waste over a long period of time. A coal spoil heap in Gloucestershire contained scattered large trees in an area undisturbed for over 100 years, close to pioneer herb-grass communities where tipping had only recently stopped (Hall, 1957). Planned disturbance also takes place where a site is cleared before redevelopment (Kamp *et al.*, 2015), which in some cases does not proceed. Other disturbances are unplanned and sporadic. At urban derelict brownfields these included trampling, tipping, vehicles, fire, and service line maintenance (Small *et al.*, 2003). Mammals can considerably disturb some sites. Rabbits, badgers, and foxes burrowed in PFA mounds, with early successional plants persisting in freshly excavated spoil (Shaw, 2009b).

Disturbances differ in extent, magnitude, frequency, predictability and turnover rate (Sousa, 1984). Footpaths may create bare ground and adjust topography in vegetated areas (Morrocco and Ballantyne, 2008). This effect is more pronounced with vehicle disturbance, forming wheel ruts (Hambleton and Drescher, 2009). Intensive vehicle activity may completely remove vegetation, and compact or displace soil across large areas, depending on the substrate (Schlacher and Morrison, 2008). Fire affects vegetation succession, providing

gaps for germination of annual and ruderal species (Velle and Vandvik, 2014). Site clearance resets succession and grading simplifies topography, at least until construction begins. Some developments may include early successional habitat creation, including 'biodiverse' green roofs (Williams *et al.*, 2014) and brownfield landscaping (Connop *et al.*, 2016).

1.3 The conservation value of brownfields

1.3.1 Studies of brownfield invertebrates

There is extensive published literature documenting the invertebrate fauna at European brownfields, summarised in Table 1.1. This repeatedly demonstrates the function of post-industrial and operational sites as analogues for threatened semi-natural habitats. Most research refers to the Czech Republic. In a comparison with peat bog habitats and agricultural ponds, 50 of 54 Odonata species occurred at mine subsidence pools (Dolný and Harabiš, 2012). 101 invertebrate species with national conservation status were recorded at PFA deposits, including 34 aculeate Hymenoptera and spiders classified as specialists of continental drift sands (Tropek *et al.*, 2014). Sand and gravel pits supported 81 Arachnida, 68 Coleoptera and five Orthoptera associated with continental drift sands, xeric steppes and exposed riverine sediments (Heneberg and Řezáč, 2014; Heneberg *et al.*, 2016b). Invertebrate assemblages associated with galls of *Lipara* frit flies (Chloropidae) in *Phragmites australis* were similar at post-industrial sites and exposed riverine sediments, including red-listed Arachnida, Coleoptera, Diptera, Lepidoptera and Hymenoptera (Bogusch *et al.*, 2016). Limestone quarries also served as secondary habitats for regionally rare butterflies of early successional steppe grasslands and scrub (Beneš *et al.*, 2003).

Relatively few studies deal with UK sites. The value of industrial wastes for vascular plant conservation was first recognised in the 1970s (Lee and Greenwood, 1976; Greenwood and Gemmell, 1978). Invertebrate interest was documented later. In England, post-industrial and urban brownfields produced records of Coleoptera associated with sand, chalk, riverine sediments, unmanaged grassland, and bare wet ground, including 46 nationally rare and scarce species (Eyre *et al.*, 2003). Most faunal surveys are only available as grey literature. Over 7,580 species were identified at man-made sites around

Table 1.1 Taxonomic scope and invertebrate species richness reported in published studies at European brownfields.

Location	Site classification	Invertebrate groups + number of species	Source
Freiburg, Germany	Urban derelict sites	Hymenoptera – Apoidea (122)	Kratochwil and Klatt (1989)
Leipzig, Germany	Lignite spoil heaps (3)	Coleoptera (203)	Brändle *et al.* (2000)
Lower Lusatia, Germany	Brown coal mining spoil heaps	Arachnida (215)	Mrzljak and Wiegleb (2000)
Moravia, Czech Republic	Limestone quarries (21)	Lepidoptera – Butterflies (82)	Beneš *et al.* (2003)
England	Calcium carbonate dump, colliery spoil heaps (5), railway land (3), sand pit (2), iron works (2), ex-industrial (2), roadside (1)	Coleoptera (473)	Eyre *et al.* (2003)
London, England	Pulverised fuel ash dumps (3)	Collembola (21)	Shaw (2003)
West Midlands, England	Rubble, ballast + brick (26)	Coleoptera – Carabidae (63)	Small *et al.* (2003)
Lower Lusatia, Germany	Brown coal mining spoil heaps	Heteroptera (138); Hemiptera – Auchenorrhyncha (92); Coleoptera (518); Arachnida (263); Orthoptera (35)	Bröring and Wiegleb (2005)
Lower Lusatia, Germany	Brown coal mining spoil heap	Heteroptera (127)	Bröring *et al.* (2005)
Sokolov, Czech Republic	Brown coal mining spoil heaps	Hymenoptera – Formicidae (20)	Holec and Frouz (2005)
Bremen & Berlin, Germany	Urban derelict sites	Hemiptera – Auchenorrhyncha (194)	Strauss and Biedermann (2006)
Limousin region, France	Leucogranite quarries (6)	Orthoptera (21)	Picaud and Petit (2007)
SW Czech Republic	Limestone quarry (1), granulite quarries (2)	Coleoptera – Carabidae + Arachnida	Tropek *et al.* (2008)
Bedfordshire, England	Chalk quarry (1)	Lepidoptera – *Hamearis lucina*	Turner *et al.* (2009)
Kujawy, northern Poland	Soda ash dumps (6)	Aculeate Hymenoptera – Chrysididae (26)	Banaszak and Twerd (2010)

Location	Site classification	Invertebrate groups + number of species	Source
Bohemian Karst Protected Landscape Area, Czech Republic	Limestone quarries (3)	Orthoptera (28); Heteroptera (94); Hemiptera – Auchenorrhyncha (88); Lepidoptera (71); spiders (136); Coleoptera – Carabidae (85); Myriapoda (20); Isopoda (9); Opiliones (8)	Tropek *et al.* (2010)
Upper Silesia, SW Poland & NE Czech Republic	Coal mine subsidence pools (18)	Odonata (50)	Dolný and Harabiš (2012)
Karvina, Czech Republic	Coal mine subsidence pools (10)	Odonata (46)	Harabiš and Dolný (2012)
Northern Czech Republic	Brown coal mining spoil heaps	Gastropoda, Myriapoda, Isopoda, Heteroptera, Carabidae, Silphidae (140)	Hendrychová *et al.* (2012)
Southern Poland	Gravel pits (50)	Lepidoptera – Butterflies (71)	Lenda *et al.* (2012)
Uppsala County, Sweden	Sand pits (13)	Coleoptera (256)	Lönnberg and Jonsell (2012)
Kladno, Czech Republic	Black coal spoil heaps (5)	Orthoptera (19); spiders (100); Hemiptera – Auchenorrhyncha (59); Coleoptera (124), Heteroptera (78); Lepidoptera (49)	Tropek *et al.* (2012)
Tees Valley, England	Post-industrial sites, particularly blast furnace slag deposits	Lepidoptera	Woods (2012)
North-western Czech Republic	Lignite spoil heaps (9)	Odonata (32)	Harabiš *et al.* (2013)
Czech Republic	Sand and gravel pits (54)	Aculeate Hymenoptera (221)	Heneberg *et al.* (2013)
Brno-Líšeň, Czech Republic East Midlands, England	Limestone quarry (1) Restored landfills (9)	Coleoptera – Carabidae (43) Flower-visiting insects	Novotná and Šťastná (2013) Tarrant *et al.* (2013)
Kladno, Czech Republic	Black coal spoil heaps (11)	Lepidoptera (91)	Tropek *et al.* (2013b)
Czech Republic	Sand and gravel pits (28)	Arachnida (241)	Heneberg and Řezáč (2014)

Location	Site classification	Invertebrate groups + number of species	Source
Poland	Fly ash heap (1) + colliery spoil heap (1)	Coleoptera – Carabidae (84)	Schwerk (2014)
Polabí, northern Czech Republic	Fly ash deposits (2)	Aculeate Hymenoptera (277); Lepidoptera – moths (202); spiders (65); Diptera – Syrphidae (30); Hymenoptera – Formicidae (11)	Tropek *et al.* (2014)
Eastern Slovakia	Waste dumps (8)	Soil macrofauna (17 orders)	Baranová *et al.* (2015)
Ostrava, Czech Republic	Black coal spoil heaps (3)	Coleoptera (303)	Hodecek *et al.* (2015)
East Midlands, England	Restored landfills (9)	Coleoptera – Carabidae (37)	Rahman *et al.* (2015)
Czech Republic	Lignite spoil heap (1)	Odonata (22)	Tichanek and Tropek (2015)
Czech Republic	Post-industrial sites (15)	Araneae (14 + 10 morphospecies); Heteroptera (6 species); Hemiptera – Auchenorrhyncha (1 + 3 morphospecies); Lepidoptera (2); Sternorrhyncha (2); Coleoptera (17 + 3 morphospecies); Hymenoptera (33 + 54 morphospecies); Diptera (9 + 3 morphospecies)	Bogusch *et al.* (2016)
Czech Republic	Sand and gravel pits (28)	Coleoptera (401); Orthoptera (43)	Heneberg *et al.* (2016b)
Czech Republic	Off-road motorcycle tracks (19)	Aculeate Hymenoptera (302)	Heneberg *et al.* (2016a)
East Midlands, England	Restored landfills (9)	Pulmonata (10)	Rahman *et al.* (2016)
Greater Manchester, England	LeBlanc waste tip (1)	Megadrilacea (11)	Butt and Briones (2017)
Central Poland	Soda ash dumps (2)	Aculeate Hymenoptera - Sphecidae (64)	Twerd *et al.* (2017)

the Thames Estuary in England, including national rarities (Harvey, 2008; Robins *et al.*, 2013). Industrial processes have created important habitats in this area. 1,243 species of invertebrate were recorded from PFA deposits at West Thurrock between 1996 and 2005, including 35 Red Data Book and 116 Nationally Scarce species (Harvey, 1995). The jumping spider *Sitticus distinguendus* is only known in the UK from this site, and a cement factory flue ash deposit in Kent (Harvey, 2006). The springtail *Isotoma riparia* was also recorded as new to Britain from surface layers of PFA at East Thurrock (Shaw, 2003).

Aggregate extraction has supplemented natural exposures of sand, gravel, chalk and clay, providing an artificial analogue for former habitats of the River Thames terraces (Klemm, 1996). Sixteenth century leases for chalk quarrying at Purfleet, Essex, emphasise the long continuity of this activity (Cracknell, 1968). Aculeate Hymenoptera are well represented, with 74% of the national fauna recorded in the East Thames Corridor (Penny Anderson Associates, 2005). Sandy habitats have also been created by river dredging. Since the foundation of the Thames Conservatory in 1857, the River Thames has been dredged to maintain and improve commercial navigation (Port of London Authority, 2014). Most sites at the inner Thames estuary now use water injection dredging but this is not always suitable and there has also been onshore disposal. There are operational sites at Rainham Marshes and Cliffe Pools, and in the 1970s there was disposal at Canvey Wick. 1,273 species of invertebrate were recorded at Canvey Wick between 1999 and 2002, including 25 Red Data Book and 107 Nationally Scarce species (Catherine Bickmore Associates, 2004).

Historically, development in London's East End reflected access to the River Thames and siting of polluting activities downwind from the city centre. Damage during World War Two, followed by de-industrialisation and economic breakdown in the 1970s, created a legacy of urban derelict land. Nationally, similar sites support fewer rarities than chemically extreme substrates (Small *et al.*, 2003), although at present the ground beetle *Brachinus sclopeta* and the weevil *Pseudoperapion brevirostre* are only recorded in two East London boroughs (Hodge, 2011; Passingham, 2013). A network of derelict sites may also be critical to landscape-scale biodiversity conservation in cities (Strauss

and Biedermann, 2008), and as a source of potential colonists for newly-created urban green infrastructure (Pankhurst, 2012).

1.3.2 Themes in brownfield invertebrate research

The most common approach in brownfield invertebrate research is to sample invertebrates within a gradient of vegetation succession. These studies emphasise the impact of ecological succession on the range of life cycles that can be supported. For springtails at PFA dumps, a small community of early successional epiedaphic species was replaced by a denser and richer community of generalists which included euedaphic species (Shaw, 2003). This was interpreted as the result of soil development creating new microhabitats as woodland developed. Ongoing succession at lignite spoil heaps produced a highly dynamic area, with a variety of habitats for Coleoptera (Brändle *et al.*, 2000). Regional conservation priority species were associated with early successional stages of bare soil and pioneer vegetation (Brändle *et al.*, 2000). Greater density of trees limited the pool of ant species at brown coal spoil heaps, compared with early successional areas (Holec and Frouz, 2005). Fewer studies examine urban derelict brownfields, although carabid species richness was also greatest at early successional sites (Small *et al.*, 2003).

The development of vegetation is strongly associated with phytophagous insect assemblages (Littlewood *et al.*, 2012). Plant and leafhopper species richness were positively correlated at brown coal spoil heaps, in addition to impacts of vegetation architecture on community structure (Bröring and Wiegleb, 2005). Leafhopper assemblages changed considerably along a successional gradient of vegetation structure at urban derelict sites, although species richness remained relatively constant (Strauss and Biedermann, 2006). Floral diversity in early to mid-successional stages can also be beneficial. Threatened Lepidoptera at black coal spoil heaps in the Czech Republic were associated with ruderal communities (Tropek *et al.*, 2013b). In Berlin, 400 plant and 100 bee species were recorded in 0.03 ha of disused railway land, including 21 oligoletic taxa (Saure, 1996). At urban derelict sites, bee species diversity was higher among mosaics of vegetation in different stages of phytosociological progression (Kratochwil and Klatt, 1989).

The association of ground-nesting aculeate Hymenoptera with fine-grained substrates is a recurrent theme. At fly ash wastes in the Czech Republic, barren deposits were more important than overgrown plots (Tropek *et al.*, 2013a). Differences in particle size and compaction also created niche specification effects at colliery spoil heaps (Srba and Heneberg, 2012). The extent to which epigeal invertebrates are influenced by post-industrial substrates is poorly understood. In the Czech Republic, 75% of Coleoptera species in sand quarries were absent from fine-grained black coal spoil and fly ash deposits (Heneberg *et al.*, 2016b). These studies were not directly comparable and experimental evidence is currently lacking. Topographic variation is a further source of physical habitat heterogeneity. Burrowing aculeate Hymenoptera were associated with vertical or near-vertical slopes in sand quarries (Heneberg, 2012). Undulating topography at mining overburden dumps created permanent and ephemeral aquatic habitats that were colonised by dragonflies (Harabiš *et al.*, 2013; Tichanek and Tropek, 2015).

Sporadic disturbances delay vegetation succession and increase habitat heterogeneity. Active limestone quarries in the Czech Republic supported xerophilous and sedentary Lepidoptera of sparsely vegetated semi-natural habitats (Beneš et al., 2003). Low intensity ground disturbance at off-road motorbike circuits was associated with increased species richness and abundance of red-listed aculeate Hymenoptera in the Czech Republic (Heneberg *et al.*, 2016a). For species that require multiple resources, a small-scale mosaic provides the microhabitats necessary for persistence within the landscape (Saure, 1996). Activity ranges are poorly understood for most species. The minimum mean distance shrill carder bee *Bombus sylvarum* workers travelled from nest to forage patch was calculated as 231 ± 58 m (Connop *et al.*, 2011). Maximum foraging distances for *Hylaeus punctulatissimus* (a small solitary bee) were 1,100 m, although 50% of females did not forage beyond 225 m (Zurbuchen *et al.*, 2010). Other species may have smaller activity ranges. A study of *Formica cunicularia*, a heathland ant, estimated that very few workers foraged more than 10 m from the nest (Kessock-Philip *et al.*, 2004).

Other studies compare unrestored and reclaimed brownfields. Undirected succession and forestry reclamation increased overall species richness at

brown coal spoil heaps, although assemblages were not assessed for conservation value (Hendrychová *et al.*, 2012). At black coal spoil heaps, unrestored sites hosted most Arachnida, Orthoptera, Coleoptera, Heteroptera and Lepidoptera of conservation concern, which were nearly absent from reclaimed plots (Tropek *et al.*, 2012). Most threatened bees, wasps and spiders were recorded at unrestored PFA deposits, although some aculeates also occurred in areas restored to grassland (Tropek *et al.*, 2014). Spider diversity increased with habitat age at sand quarries, and conservation interest was greatest 6-15 years after mining (Heneberg and Řezáč, 2014). Reclaimed sites lacked many species present in unrestored successions but did also host critically endangered taxa. The study did not focus on scrub encroachment, although it was noted that most red-listed species were absent from areas overgrown by trees.

The surrounding landscape could affect brownfield invertebrate community composition. Forest cover around sand quarries was positively correlated with psammophilous and forest-associated Coleoptera, and negatively associated with open habitat species (Lönnberg and Jonsell, 2012). For Lepidoptera in limestone quarries, xerophilous species decreased with site age and sedentary species decreased with forest cover, while both groups were positively associated with the surrounding area of steppe (Beneš *et al.*, 2003). Species richness and diversity of butterflies at gravel pits increased with proximity of grasslands, and decreased in relation to coniferous plantations (Lenda *et al.*, 2012). For leafhoppers and grasshoppers at urban derelict sites, many species responded positively to high or intermediate density of grassy brownfields within 25–125 m (Strauss and Biedermann, 2006). Urban carabid beetle assemblages were principally related to site age, substrate type and vegetation community, rather than brownfield density or linear connectivity, although measured at scales of 100–5,000 m (Small *et al.*, 2006). Four butterfly species were also not affected by proximity to other urban derelict sites (Angold *et al.*, 2006).

The potential influence of site size has rarely been examined. For carabids, the area of sand quarries was the major influence on species composition (Lönnberg and Jonsell, 2012). The main difference was that most psammophilous species were associated with medium or large (5–18 ha) rather than smaller workings (0.36–0.7 ha). Results for other groups are inconsistent,

as habitat area did not affect species richness or abundance for butterflies in gravel pits (Lenda *et al.*, 2012). A few studies address species ecology at post-industrial sites. The butterfly *Hamearis lucina* was found to prefer dense patches of larger foodplants (*Primula*) growing in sheltered locations (Turner *et al.*, 2009). There were frequent extinctions within a metapopulation of the dragonfly *Leucorrhinia pectoralis* at mine subsidence pools but these were not related to size and spatial isolation of single habitats (Harabiš and Dolný, 2012).

Bare ground, structural heterogeneity, and floral diversity are also valuable in semi-natural habitats such as chalk grassland and heathland (Mortimer *et al.*, 1998; Morris, 2000; Cameron and Leather, 2012a). Brownfields differ in the development and persistence of these components. Nutrient-poor and contaminated substrates are slow to re-vegetate, which means that flower-rich habitats can remain open for decades without regular management. Early successional communities on PFA persisted for up to 15 years before scrub colonised (Shaw, 1992) and for a similar period on compacted rubble (Small *et al.*, 2003). Succession was delayed for 50 years on highly acidic tertiary sands at lignite dumps (Brändle *et al.*, 2000). The absence of cutting or grazing provides continuity of plant structural resources, including aerial stems, foliage, buds, flowers, fruits and seeds (Harvey, 2000; Harvey, 2007). Leaf mining Lepidoptera were demonstrated experimentally to be highly sensitive to grazing management, predominately through effects on foodplant species, and also plant structure (Sterling *et al.*, 1992).

1.3.3 Other research about brownfields as important landscape components

In comparison with biodiversity conservation, other ecosystem services functions of brownfields are less well-researched. Potentially important services include water attenuation, carbon sequestration, greenspace access, climate, and air pollution regulation (The Economics of Ecosystems and Biodiversity, 2010). This contrasts with general planning discourses, including the definition of brownfield land as requiring restoration to ‘beneficial’ economic use (Oliver *et al.*, 2005). Some caution is needed, as maximising one service may not optimise overall ecosystem services production (Windhager *et al.*, 2010). From this perspective, tree planting on brownfields to increase above-ground carbon storage (Davies *et al.*, 2011) could lead to a reduction in regionally valuable

biological diversity. This suggests that multi-disciplinary research is needed, to encompass the range of ecosystem services associated with different sites and management practices (Robinson and Lundholm, 2012).

Public attitudes to the brownfield landscape and associated wildlife are potentially important. Many urban residents make use of brownfields and have ideas about how these should develop in the future, which could strengthen the case for conservation. Possible benefits include recreation value, mental respite, noise reduction, green job creation and environmental connection (Anderson and Minor, 2017). A study in Brisbane, Australia and Sapporo, Japan suggested that most respondents (80%) knew of informal green space in their neighbourhood (Rupprecht *et al.*, 2015). This was visited more frequently than formal urban greenspace because it was closer, featured more diverse flora and fauna, and had no usage restrictions (Rupprecht *et al.*, 2015). Almost 30% of respondents reported using informal urban green space as teenagers, to observe wildlife or plants, increasing to 40% for childhood (Rupprecht *et al.*, 2016).

Cultural studies value marginal wastes as wild space, offering experiences and visual catharsis in a new framing of familiar urban surroundings (Farley and Roberts, 2012; Chell *et al.*, 2013). Fifteen urban habitats classified by broad vegetation type were also thought to have medium to high social value, including amenity, educational and scientific uses (Tucker *et al.*, 2005). Landschaftspark Duisburg-Nord in Germany has been presented as a post-industrial site that produces and enriches the culture of surrounding communities (Langhorst, 2004). Different successional stages of spontaneous vegetation were universally accessible and some of the remaining industrial structures were used for outdoor recreation. Social change was evident in the daily use of the park by local residents and participation by former site workers as tour guides. Other sites are used for informal community recreation (Shoard, 2002) or as educational resources (CPIE Chaîne des terrils, 2014).

Divergent perceptions of brownfield vegetation have been reported. Respondents in Blois and Tours, France, expressed a preference for grassy wastelands, in comparison to pioneer vegetation or shrub-dominated sites (Brun *et al.*, 2017). Respondents in Leipzig and Bremen, Germany, also responded less positively to sites characterised by extensive surface sealing and low

vegetation cover (Mathey *et al.*, 2016). Positive perceptions increased with more persistent ruderal and tall herbaceous vegetation, although managed, park-like areas were preferred to unmanaged succession. Differing attitudes may relate to regional variations in the density and persistence of brownfields, and proximity to residential areas (Levi and Kocher, 2006; Nassauer and Raskin, 2014). Recognising differing perceptions of brownfields could minimise conflicts between stakeholders, and promote the social acceptability of management activities (Mathey *et al.*, 2016).

1.4 Assessment of previous brownfield research

Conservation of individual brownfields might be unnecessary if there was sufficient transient habitat to support invertebrate metapopulations. Simulation modelling of 38 plant and 43 insect species at urban brownfields in Germany predicted that species richness would decline markedly after 20 years, and by half within 50 years (Kattwinkel *et al.*, 2011). To sustain a dynamic landscape of different successional stages, an average plot age of 15 years was recommended, based on 50-60% open space (Kattwinkel *et al.*, 2011). It is important that models are representative of the actual system. In 70 United States cities, on average only 15% of previously developed land (PDL) was vacant (Pagano and Bowman, 2000). Uncertainty also arises where there are regional variations in building turnover. In London, there was a 38% decrease in the amount of vacant and derelict PDL between 1998 and 2005, compared with a national fall of 5% (London Development Agency, 2007). This suggests that effective conservation management of existing sites is important, particularly for substrates that are no longer created, or rarely deposited.

There are limitations in the application of existing brownfield research to determine appropriate management of Open Mosaic Habitat. It is possible to make inferences about large-scale and long-term processes from studies within successional gradients. Data can be collected in one field season, only constrained by the variables available in the environment. Despite these advantages, it is necessary to rely on correlation to infer causal relationships, even if this is based on plausibility and consistency (Woodcock *et al.*, 2015). It is also unlikely that sites share a common environmental history. More critically, the importance of a habitat mosaic including early successional stages is well established but knowledge of how to maintain this as a management objective

is lacking. This has led to proposals to use techniques from other open habitats for brownfield conservation management. These include grazing, mowing, scrub removal, and winter burns (Saure, 1996; Beneš *et al.*, 2003; Bodsworth *et al.*, 2005; Kattwinkel *et al.*, 2011; Lenda *et al.*, 2012). Other studies have suggested planned disturbances, without addressing the comparative effectiveness of techniques to implement these (Smith *et al.*, 2014).

Even with appropriate management, conservation of isolated core sites is problematic (Simberloff and Wilson, 1969; Brown and Kodric-Brown, 1977; Kruess and Tscharntke, 1994; Saunders *et al.*, 1991). These may be inaccessible to less mobile species, particularly as climate changes (Warren *et al.*, 2001; Honnay *et al.*, 2002; Thuiller *et al.*, 2005; Oliver *et al.*, 2017). Research also suggests that genetic diversity is affected by habitat fragmentation, although links to fitness costs are not established. For UK bumblebees, population bottlenecking and diploid male production after inbreeding have been identified for moss carder bee *Bombus muscorum* (Darvill *et al.*, 2006). *Bombus sylvarum* also exhibited lower allelic richness in UK compared with French colonies, although the effect of a limited genetic foundation was not excluded (Ellis *et al.*, 2006). Recent government policy context reflects these concerns, especially to develop a coherent and resilient ecological network (Lawton *et al.*, 2010). Mosaic habitat creation could be valuable for this but there is a lack of research about design of features, or effectiveness in general.

1.5 Threats to Open Mosaic Habitat

1.5.1 Development

The status of Open Mosaic Habitat as a Section 41 ‘habitat of principal importance’ creates an unresolved contradiction within UK planning policy, as brownfield redevelopment is actively encouraged (Shaw, 2011). There is enormous pressure for regeneration of previously developed land around the Thames Estuary. Successive governments have made the area central to London’s development as a global city, and continued economic growth of the Greater South East region (Department for Communities and Local Government [DCLG], 2006). GBP 1.86 billion was invested in East London between 1981 and 1988, through the London Docklands Development Corporation. In 1995,

the Thames Gateway Planning Framework expanded to include the area along the Thames Estuary, to Southend in Essex and Sittingbourne in Kent (DoE, 1995). This was Europe's largest regeneration project, with GBP 9 billion government investment planned (DCLG, 2007).

Re-use of previously developed land was also a core principle in Planning Policy Guidance 3: Housing (DCLG, 2000). As a result, the percentage of new housing built on PDL in England rose from 55% in 1989, to 81% in 2008 (Sinnett *et al.*, 2014). Residential developments at brownfields with high biodiversity interest included 240 ha of former chalk quarries at Chafford Hundred in Essex (Robins *et al.*, 2013), and 180 ha at Barking in London, primarily pulverised fuel ash deposits. The most recent National Planning Policy Framework expects local development orders for over 90% of suitable brownfield land by 2020, where planning permission does not already exist (DCLG, 2015). It is recognised that brownfield development could be unsuitable where PDL has high environmental value, but this is not clearly defined (DCLG, 2012). As a result, there is evidence that conservation of Open Mosaic Habitat is not being fully considered in the planning system. From 2005-2012, 51% of 198 brownfields of high or medium importance for invertebrates in the Thames Gateway had an outstanding planning permission, or were partly or fully redeveloped (Robins et al., 2013).

1.5.2 Succession

In addition to the status of Open Mosaic Habitat as a priority habitat, some UK brownfields have been declared as Sites of Special Scientific Interest. Canvey Wick was designated for nationally important invertebrate assemblages, chiefly associated with herb-rich grassland, early successional habitat, scrub edge, and brackish coastal wetlands (English Nature, 2005a). A PFA lagoon at West Thurrock designated for overwintering wading birds and wildfowl (English Nature, 1991) was later found to support many threatened invertebrates (Harvey, 2005). Other brownfield nature reserves in the Thames Gateway include Untidy Industries former vehicle wrecking yard, Buckman's Hill Quarry, and Chafford Gorges. This presents different challenges, as without management open habitats are replaced by woodland, within 30 years as a primary succession on pulverised fuel ash (Shaw, 1992). Establishment of secondary woodland took up to 25 years in sand and gravel pits (Řehounková

and Prach, 2006). Eutrophication caused by nitrogen deposition in rainfall also favours more competitive plant species (UK Research on The Eutrophication and Acidification of Terrestrial Ecosystems, 2009). These changes threaten the invertebrate assemblages associated with nutrient-poor early successional stages (Schirmel *et al.*, 2015).

1.5.3 Environmental legislation

The practice of leaving bulk industrial wastes to weather and revegetate naturally has ceased in the UK (Shaw, 2011). The processes that created LeBlanc and Solvay wastes are obsolete, and strict environmental regulations now govern the deposition of unbound industrial waste ('Council directive 2008/98/EC' (2008), *Waste (England and Wales) (Amendment) Regulations 2012*). In combination with vegetation succession at existing sites, this has greatly reduced the area of infertile, early successional habitats in industrial landscapes (Shaw, 2011). Some PFA dumps have been covered by topsoil and planted with trees, to stabilise the surface ash against wind erosion, and reduce leaching of water through the deposit (Haynes, 2009; Tropek *et al.*, 2016). There are also pressures at mineral extraction sites. In the Czech Republic, there was an 82% decrease in the number of occasionally-quarried sandpits from 1992–2012 (Heneberg *et al.*, 2016b). National legislation in this period required restoration of former quarry lands to agriculture or forestry (Heneberg *et al.*, 2013).

1.6 Research overview

This research was established as a partnership between the University of East London and Buglife, a third sector charity dedicated to the conservation of UK invertebrates. It focused on management interventions at different ecological scales, which have been central to recent UK conservation strategy (Lawton *et al.*, 2010). All the research locations were within the Thames Gateway area (Figure 1.2). Two studies were based at Canvey Wick, a core site for brownfield nature conservation. This was selected as Buglife is involved in a management agreement for the site, and discussions identified a lack of information in scientific literature about brownfield conservation management techniques. Appropriate management of core areas is also important as the population size and resilience of individual species is likely to be higher at larger sites

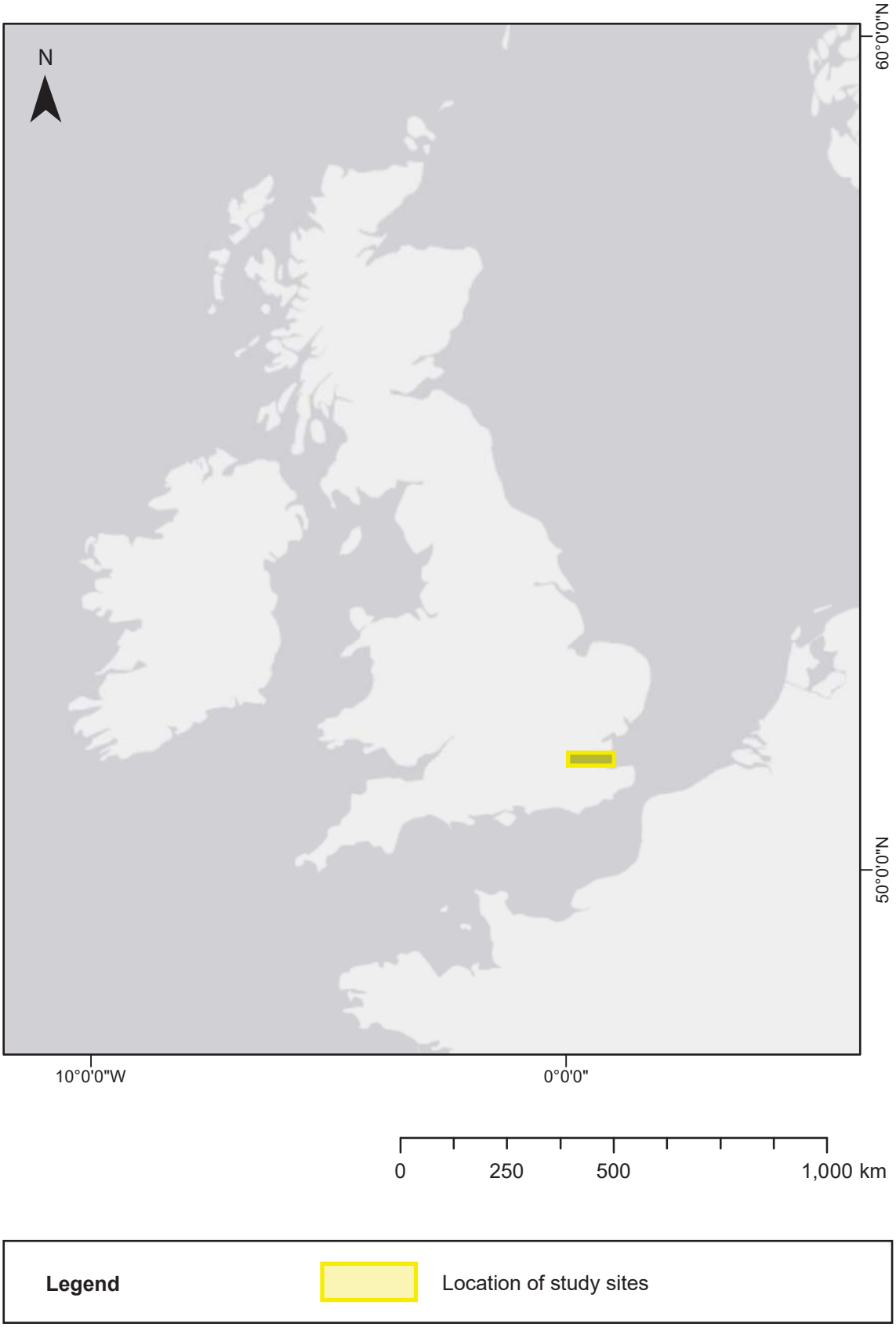

Figure 1.2 Location of study sites within Thames Gateway area of South-east England. Basemap source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS user community.

(Woodcock *et al.*, 2015). The first study examined the potential to reset succession where open habitat had been replaced by secondary woodland. The outcomes for community assembly were assessed in terms of taxonomic composition (Chapter 2) and functional traits (Chapter 3). The second study investigated comparative effectiveness of two ground disturbance techniques, where wet grassland had stabilised vegetation succession (Chapter 4).

Although designated sites have made a valuable contribution to the preservation of species, invertebrate declines have continued (Ewald *et al.*, 2015; Hallmann *et al.*, 2017). Wildlife habitats have become more fragmented and isolated, suggesting that a broader approach is required (Lawton *et al.*, 2010). The final two chapters therefore examined small scale habitat creation in the wider landscape, with a view to supplementing and connecting habitat on brownfield sites with existing conservation protection. There is a particular need to soften the matrix of farmland, which forms a much larger land area compared with protected sites (Lawton *et al.*, 2010). In the third study, aggregate introduction was trialled as a technique for creation of brownfield mosaic habitat in unmanaged grassland (Chapter 5). The fourth study investigated provision of artificial breeding habitat for ground and cavity nesting aculeate Hymenoptera on roof tops in London (Chapter 6). Green roofs have been suggested as surrogates for ground-level brownfield habitat, although evidence is lacking (Williams *et al.*, 2014). Even biodiverse roof designs might represent a partial habitat (Westrich, 1996), with suitable conditions for foraging but not nesting. A mosaic approach to habitat creation suggests that provision of rooftop nesting opportunities could facilitate populations of aculeate Hymenoptera.

1.6.1 Canvey Wick

The study site for habitat management experiments in this research was Canvey Wick in Essex (Figure 1.3), immediately west of the Canvey Island urban area (51°31'19"N 0°32'38"E). This area was formerly part of an extensive network of coastal grazing marshes (Williams and Hall, 1987). In 1972, Occidental Petroleum began construction of an oil refinery, spreading sandy dredgings from the Thames to raise the land level across almost 100 hectares. Access roads and oil storage tanks were installed, until work was halted in 1975, following the 1973-4 oil crisis (Issawi, 1978). The site was abandoned and with no further management, a varied habitat mosaic developed. Later

Figure 1.3 Location map for Canvey Wick Site of Special Scientfic Interest, Essex. Basemap source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS user community. Contains public sector information licensed under the Open Government Licence v3.0.

ecological surveys revealed nationally outstanding invertebrate assemblages, particularly associated with disturbed bare ground, herb-rich grassland, scrub edge, and brackish habitats (Catherine Bickmore Associates, 2004). As a result, 93.2 ha at Canvey Wick were designated as a Site of Special Scientific Interest (English Nature, 2005a). There created a legal obligation to maintain the complex mosaic of early successional habitats (English Nature, 2005b).

The site is now managed by a partnership of third-sector organisations. The Land Trust is a charity that aims to facilitate sustainable management of open spaces, through long-term financial investments, appointing local agents to design and deliver management plans. At Canvey Wick, an endowment was provided by the Homes and Communities Agency, as the previous landowner (Royal Society for the Protection of Birds [RSPB], 2012). Local management is the responsibility of Buglife, and the RSPB. Buglife is committed to provide specialist management advice for the site. The RSPB is the largest UK nature conservation charity, and is responsible for practical management at Canvey Wick. This research represents a first step to provide a sound evidence base for this collaboration (Lawton *et al.*, 2010). The key role of the site for experimentation with innovative management techniques is further recognised by Natural England, the non-departmental public body presently responsible for the natural environment in England (English Nature, 2005b).

1.7 Methodology

1.7.1 Invertebrates as bioindicators

The scope and taxonomic coverage in an ecological study is determined by the research objectives, as well as time and resources available for monitoring and identification. It is important to consider invertebrates, as these do not necessarily respond to management in the same way as plants or other organisms (Babin-Fenske and Anand, 2010). A typology of terrestrial invertebrates as indicators separates environmental, biodiversity, and ecological applications (McGeoch, 1998). Environmental indicators respond predictably to environmental change or disturbance (Cortet *et al.*, 1999). Biodiversity indicators include charismatic flagship taxa, or umbrella species requiring conservation of large land areas. These may also have an important role in supporting ecosystem functions but can be inconsistent as surrogates for

biological diversity (Andelman and Fagan, 2000). Ecological indicators represent the effects of environmental change on biotic systems. In this study, invertebrates were analysed as ecological indicators of habitat management effects, and intrinsic objects of conservation concern (McGeoch, 1998).

The effectiveness of invertebrates as ecological indicators has been established in many situations. These include responses to ground disturbance (Pedley *et al.*, 2013), grazing (Dennis *et al.*, 2015) and vegetation succession (Schirmel *et al.*, 2015). Most arthropods have short reproductive cycles and high population turnover. This may offer greater sensitivity to environmental change than longer-lived vertebrates, or plants with dormant propagules. Declines for UK butterflies exceeded losses of plants or birds by one to two orders of magnitude in 40 years (Thomas *et al.*, 2004). Community composition is commonly related to environmental factors, but interference and resource competition may also structure assemblages (Connell, 1983). Reduced abundance of the halophilic spider *Pardosa purbeckensis* on salt marshes invaded by the grass *Elymus athericus* was linked to competition for space and food with non-coastal lycosids (Pétillon *et al.*, 2005). Manipulative experiments are required to test for interspecific competition, but it is difficult to reconcile a species-specific approach with community-level analysis. In this research, the potential effects of competition are recognised but were not investigated further.

The spatial and temporal dynamics of species-rich communities are critical to understand the effects of environmental change on biodiversity (Brooks *et al.*, 2012). To reduce the complexity of faunal assemblages, invertebrate research often focuses on a single group. This may not be sufficient to determine appropriate conservation management, as responses can vary for different groups (Cameron and Leather, 2012b). These contrasts have been shown for ground-active beetles (carabids and staphylinids) (Fournier *et al.*, 2015), and spiders (Pedley *et al.*, 2013; Fournier *et al.*, 2015). Morphospecies or lower taxonomic resolutions have been proposed, as an alternative to increase the scale of biological monitoring (Pik *et al.*, 2002). This is unnecessary in the UK, where many invertebrate families are well-described, and taxonomically stable. Even if species-level taxonomy is more difficult, a composite focal group can increase the number and type of perceived environmental responses (Hammond, 1994). A high level of taxonomic resolution will also allow

reappraisal of results, if data are available later linking specific taxa to ecosystem services.

1.7.2 Invertebrate classification for analysis

Although the species is the fundamental unit in this research, it is difficult to assess conservation management outcomes solely from species richness. Alpha diversity indices incorporate species richness, and the abundance distribution of an assemblage (Magurran, 2004). These are independent from the number of individuals sampled but ecological effects cannot be readily interpreted. In the present study, species were instead classified as part of a target management community. These were likely to be found at sites with Open Mosaic Habitat in the Thames Gateway, although excluding eurytopic species. A potential limitation with this type of classification is that matching microclimatic and edaphic conditions may be found in different habitats, according to geographical location (Lott, 1998). This can have implications for the range of habitat structures occupied by a species in different regions. As an example, the carabid *Bembidion lunatum* was confined to river channels in northern Norway, but also occurred along roadsides and at gravel pits in central districts (Andersen, 1983).

This can affect the local classification of species, making it more difficult to compare studies. The abundant carabid *Bembidion quadrimaculatum* was classified as a dry, open habitat species at brownfields in the West Midlands (Small *et al.*, 2003) but excluded from a target management group in the Norfolk Brecks as part of an open-mesic community (Pedley *et al.*, 2013). Other classifications include the ecological amplitude for Dutch carabids, calculated based on the range of habitats occupied (Turin *et al.*, 1991). These were selected a priori and may not be evenly distributed along natural environmental gradients of importance to invertebrate communities. An alternative definition in a UK context could include preference for early successional habitat and conservation status, but the species pool in the present research was too limited for this. As a result, the grouping chosen must be regarded as approximate, but this was still useful for local assessment of conservation management.

1.8 Sampling methods

1.8.1 Pitfall traps

The objective in this research was to detect differences between experimental treatments based on invertebrate community composition. Pitfall traps have been widely used to sample ground-dwelling invertebrates, including the study of activity patterns, population density, relative abundance and habitat associations (Woodcock, 2005). The main advantage is the potential to collect large standardised samples, continuously at multiple locations. Other practical benefits are low cost, and the limited time needed to deploy and service traps. Against this, possible sources of trapping bias must be considered. Constant biases include trap size (Work *et al.*, 2002), shape (Obrist and Duelli, 1996), colour (Buchholz *et al.*, 2010), preservative (Skvarla *et al.*, 2014), materials (Luff, 1975), covers (Baars, 1979; Buchholz and Hannig, 2009), and depletion effects (Digweed *et al.*, 1995). As most research uses different trapping methods depending on objectives and available materials, the main issue is that this prevents easy comparison between studies. A fully-described method can be repeated and allows assessment of factors that might influence the catch.

There are also systematic biases. One limitation is that pitfall traps particularly sample larger ground-active species, rather than the entire community of epigeal arthropods (Oxbrough *et al.*, 2005; Hancock and Legg, 2012). Sample composition is further influenced by the density of the sampled population, and trap avoidance behaviour (Topping and Sunderland, 1992; Gerlach *et al.*, 2009). The abundance of individual species also varies temporally, due to behaviours including mate searching, and dispersal of females after diapause (Penney, 1966). As a result pitfall catches do not truly reflect the relative abundance of different species, although it is possible to consider relative abundances of the same species over a complete season (Baars, 1979; Topping and Sunderland, 1992).

Limitations in providing a complete inventory of ground-dwelling species, estimates of population density, or phenological differences did not impact the objectives of this research. The main issue was the potential effect of differences in vegetation density between treatments and controls. Catches for individual species may vary depending on the resistance to horizontal

movement exerted by different types of ground cover (Greenslade, 1964). The activity density of *Pterostichus melanarius*, a large carabid, was significantly higher in bean than wheat crops, despite similar population density (Thomas *et al.*, 2006). Larger catches of abundant ants were reported after experimental manipulation of grassland within 30 cm of traps, with relative abundance variably affected (Melbourne, 1999). Most species with less than 10 individuals were not caught by traps in unmodified habitat, although the study only ran for two weeks and the extent to which species were responding to the habitat manipulation is unclear.

Pitfall trap catches of carabid beetles from a mosaic of bare ground and chalk grassland were compared with a closed sward over 10 weeks (Cleverly, 2002). Species in the genera *Pterostichus* and *Amara* associated with unmanaged grasslands were more frequent in the closed sward. This implies that pitfall sampling detected an actual difference in carabid species because larger numbers would otherwise be expected at the disturbed site. Community composition from pitfall traps and hand searching was also similar in a forestry plantation stand, clear-fell, and ride margin (Lin *et al.*, 2005). While recognising the existence of sampling bias, pitfall traps remain the most common method to collect data about invertebrate habitat associations over large spatial and temporal scales. In this research, it is acknowledged that abundance data reflects a combination of relative activity and population density of individual species, which is frequently referred to as activity density (Thomas *et al.*, 1998).

1.8.2 Alternative sampling methods

Other collecting methods were considered. Destructive sampling of the habitat for extraction was not possible where repeated samples were needed from a limited area. Quadrat sampling has been used to calculate absolute density for cursorial spiders but this covers limited time and introduces temporal stratification (Uetz and Unzicker, 1975). In a study of moorland Lepidoptera, sampling could not be completed at all sites simultaneously, and in subsequent analysis the strongest environmental variable was the stage of the season when samples were collected (Littlewood *et al.*, 2006). In habitats with extensive bare ground it is also likely that many quadrats would contain few or no specimens (Lott, 1998). While acknowledging that quadrats are suitable to sample population density, including slow moving and inactive species, pitfall trapping

provides an effective description of the overall community. Pitfall traps caught 94% of a combined 130 spider species, compared with 41% for visual searching and 25% for sweep netting (Churchill and Arthur, 1999). In sparsely vegetated dry river beds, 80% of taxa were unique to pitfall traps, and 4% to quadrat samples (Corti *et al.*, 2013).

Samples could be standardised by time rather than area. In a comparison of community composition in pitfall traps and timed hand searches, smaller differences were attributed to collection method than environmental or seasonal factors (Lott, 1998). Night-active species could be underrepresented in samples from hand collection, as demonstrated for riverbanks (Andersen, 1995). Another study found no effect of collection method on diel activity, although this relied on splashing water from rivers on areas of bare ground (Lott, 1998). Hand collection was not used in the present research, because of difficulty in accessing the study sites at night, and the lack of water to assist with searching. Suction sampling is an alternative method to collect invertebrates that are more active in the field layer, including phytophagous Coleoptera (Woodcock *et al.*, 2008). This would provide a more comprehensive assessment of the invertebrate community, and might also correct for some aspects of pitfall trap bias. In grasslands, larger-bodied beetles and spiders were captured more frequently by pitfall traps, in comparison with combined D-Vac and swish net samples (Standen, 2000). Suction sampling could also be standardised to area, but was not possible on the bare, sandy substrates in the present research.

Chapter 2 – Canvey Wick restoration: taxonomic analysis

2.1 Introduction

Semi-natural grasslands are traditionally managed as open habitats by cutting or grazing (Schirmel *et al.*, 2015). These techniques can support species-rich invertebrate communities, although some life-cycles are disrupted by removal of plant structural resources (Volkl *et al.*, 1993). Open Mosaic Habitat lacks routine management, with an extended early successional phase maintained by a combination of low-nutrient substrates and sporadic disturbance (Small *et al.*, 2003). This encourages the development of architectural complexity in vegetation, which is a key driver for invertebrate occurrence (Woodcock and Pywell, 2009). A single host plant may support multiple phytophagous species, utilising resources including roots, stems, leaves, buds, flowers and seeds (Harvey, 2000; Harvey, 2007). Seed set is also important for graminivorous carabids, with some species particularly occurring in anthropogenic habitats with ruderal vegetation (Andersen, 2000). Niche differentiation among predatory taxa was greater in architecturally complex swards (Woodcock *et al.*, 2007). Vegetation tip height diversity increased the potential for construction of different web types among spiders (Traut, 2005). Tussock forming grasses were also important habitats for carabid and staphylinid beetles (Luff, 1966).

While brownfields may remain open for decades without management (Shaw, 1992), most early successional habitats are eventually replaced by secondary woodland (Řehounková and Prach, 2006; Schadek *et al.*, 2009). The mean rate of scrub increase in calcareous grassland was 1,096 m^2 per ha after 10 years, or 10.96%, assessed from decadal aerial photography (Redhead *et al.*, 2012). This did not address potential non-linear annual trends, and the development of brownfield scrub is also likely to vary for different substrates. The proximity of seed sources was critical in the development of heathland scrub (Manning *et al.*, 2006). Wind-dispersed genera include *Betula*, *Salix*, *Alnus* and *Pinus* (Mortimer *et al.*, 2000). For fleshy-fruited shrubs, birds were the major vector of seed dispersal (Kollmann, 1995). Little information is available about soil development on marine dredgings but at grey dunes and heaths, availability of nitrogen, calcium, aluminium and potassium increased with succession to birch forest (Mantilla-Contreras *et al.*, 2011). This was also associated with decreased temperature at ground level, and higher humidity (Mantilla-Contreras

et al., 2011). Floral diversity was reduced by scrub invasion in grasslands, with increased shade and soil organic matter favouring shade-tolerant and competitive species (Davies and Waite, 1998).

Scrub can have high conservation value, particularly in a mosaic with grassland which provides a variety of niches, food sources, and shelter for plants and animals (Mortimer *et al.*, 2000). Abundance and species richness of bumblebees and butterflies increased along field margins in the shelter of hedgerows, as well as floral abundance (McCracken and Tallowin, 2004). Some invertebrates are associated with specific vegetation structures. Butterflies benefitted from development of scattered *Prunus spinosa* in old fallow grassland, although specialist species were completely lost after trees formed a closed canopy (Balmer and Erhardt, 2000). The UK conservation priority moth *Aleucis distinctata* also preferentially laid eggs on *Prunus spinosa* suckering into grassland (Butterfly Conservation, 2015). Scattered trees alter microclimate (Manning *et al.*, 2006), nutrient cycling (Rossetti *et al.*, 2015), trap leaf litter (Lindsay and Cunningham, 2009), and enhance structural complexity for decomposers, detritivores and associated predators. Decomposing organic matter and woody debris may also favour the development of fungi, and mycetophilous invertebrates (García-Tejero and Taboada, 2016).

The process of ecological succession profoundly affects the brownfield invertebrate fauna (Hodecek *et al.*, 2015). A mosaic of all successional stages was required to maintain the regional species pool of leafhoppers at urban derelict sites in Germany (Strauss and Biedermann, 2006). Even if diversity does not change along successional gradients, a possible concern is the loss of conservation priority species. Tree-invaded sites were unsuitable for carabid assemblages of dry grasslands and heathlands (Buchholz *et al.*, 2013). A reduction in disturbance effects was also associated with the loss of specialised spiders at grey dunes (Bonte *et al.*, 2006). This suggests that scrub management or clearance is required to maintain a mosaic of open habitat elements (Staley *et al.*, 2015). There have been limited attempts to restore the botanical interest of industrial wastes from secondary woodland. At two PFA dumps where *Dactylorhiza* orchid colonies were overgrown by scrub, all vegetation and organic soil horizons were mechanically removed (Shaw, 1998). Legumes, other forbs and graminoids colonised but without the halophytes

characteristic of newly deposited PFA, and with no evidence of orchid establishment within five years (Shaw, 1998).

There is no comparable information about management techniques for early successional invertebrates, on brownfields set aside for conservation. Scraping substrate has been recommended (Shaw, 2011) but detailed knowledge of subsequent invertebrate colonisation is lacking. There are also potentially conflicting requirements to maintain early successional stages, and allow associated invertebrate communities to develop. Examining species turnover immediately after scraping for multiple taxonomic groups is critical to understand this. Scraping has been studied in heathland and dry grassland but monitoring excluded spiders (Schirmel, 2010; Cameron and Leather, 2012a), staphylinids (Pedley *et al.*, 2013), or commenced more than one year after disturbance (Ödman *et al.*, 2011; Olsson *et al.*, 2014). This suggests that stenotopic species are likely to benefit from scrub removal, although it cannot be presumed that a successful technique in one habitat can be directly transferred to another. There is also little evidence of the potential management legacy after scraping, including colonisation of bare ground in a scrub matrix by seedlings of woody shrubs, and the potential for invasion of alien plants.

This study evaluates the restoration of an area by removing scrub and trees, at a site previously used for onshore disposal of marine dredgings. The development of invertebrate communities was monitored in a single year, in different areas with all vegetation and organic matter removed in the previous two winters. It was expected that assemblages of carabids, staphylinids and spiders would be influenced by the time since clearance. To explore this, the following questions were asked. (1) Do restored plots support greater abundance and richness of stenotopic species than control or reference plots? (2) Do restored plots increase in similarity to reference plots with time since disturbance? (3) Is it possible to identify indicator species for the different stages of succession? (4) How does habitat structure develop in the two years following clearances?

2.2 Methods

2.2.1 Experimental design

The location for this research was Canvey Wick in Essex (51°31'19"N 0°32'38"E), a post-industrial site created in the early 1970s by disposal of marine dredgings (Figure 1.3). By 2013 there were extensive areas of scrub, primarily *Betula pendula* and also *Hippophae rhamnoides*, *Populus alba*, *Crataegus monogyna* and *Prunus spinosa*. Due to the loss of open habitat, a restoration management plan was agreed for some areas with a closed canopy of young trees. A continuous 1 ha block of scrub was cleared in November 2013 (‘phase one’, Figure 2.1). Leaf litter was scraped and tree stumps dug out with an excavator in May 2014, leaving bare sand. This process was repeated in an adjacent 1 ha area in November 2014 and May 2015 (‘phase two’, Figure 2.2). The restoration areas and unmanaged controls (Figure 2.3) were compared as a time series in 2015. Data were also collected at reference plots, maintained as a mosaic of bare ground and early successional vegetation by disturbance from rabbits *Oryctolagus cuniculus* (Figure 2.4). There were three monitoring plots in the phase one restoration, six in the phase two restoration, six in two blocks of scrub with a closed tree canopy (three in each), and five reference plots (Figure 2.5). The blocks of scrub were to the north, west and east of the restoration areas, and the reference plots were also within 100 metres of these.

2.2.2 Invertebrate sampling

At each monitoring plot, invertebrates were captured in pitfall traps separated by 2 m, in two rows of three. Traps were polypropylene cups, 8.0 cm diameter and 10.5 cm deep, with 50 ml preservative (50% propylene glycol and 50% distilled water) and a drop of detergent to break the surface tension. To reduce capture of small mammals and reptiles, all traps were covered with 13 mm x 13 mm wire mesh. Traps were open for nine-day periods from 11-19 June, 13-21 July, and 11-19 August. The sampling programme provided adequate seasonal coverage, including key periods for invertebrate activity in early successional habitats (Lott, 1998), and minimised potential depletion effects (Digweed *et al.*, 1995). Samples were preserved in 70% industrial methylated spirits.

Figure 2.1 Phase one restoration area at Canvey Wick, June 2015. Just over a year after scraping, there was extensive regeneration from rhizomes of *Phragmites australis*. There was patchy bare ground among the vegetation.

Figure 2.2 Phase two restoration area at Canvey Wick, May 2015. A fortnight after scraping, there were large areas of bare ground, with some early regeneration from *Phragmites australis* rhizomes.

Figure 2.3 Scrub at Canvey Wick with *Betula pendula* and *Salix* species, June 2015. Areas of closed canopy scrub were monitored as controls, for comparison with the outcomes of habitat restoration.

Figure 2.4 Early successional habitat at Canvey Wick, October 2014. Areas with a mosaic of bare ground and pioneer vegetation were monitored as reference plots, to assess the outcomes of habitat restoration.

Figure 2.5 Location of monitoring plots in 2015 for assessment of restoration management at Canvey Wick, Essex. Control plots were within closed canopy scrub. Phase one restoration was cleared to bare sand in May 2014. Phase two restoration was cleared to bare sand in May 2015. Reference plots contained a mosaic of bare ground and early successional vegetation. Basemap source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS user community.

2.2.3 Invertebrate classification

Carabid and staphylinid beetles (Order: Coleoptera, Families: Carabidae, Staphylinidae) and spiders (Order: Araneae) were selected for study in this research. These groups contain diverse life-histories and ecological preferences, and epigeal species are effectively sampled by pitfall traps. Adult carabid beetles were determined to species from Luff (2007). Adult staphylinid beetles were determined to species from Freude *et al.* (1974), Lott (2009), Lott and Anderson (2011), and Assing and Schülke (2012). Adult spiders were determined to species from Roberts (1985) and Roberts (1987). Some specimens could not be identified from morphological characteristics, or genitalia. Staphylinids (89 specimens) were female *Gabrius*, *Carpelimus bilineatus*/*erichsoni*, *Carpelimus rivularis/similis*, *Anotylus sculpturatus/mutator*, *Gyrohypnus angustatus/wagneri*, and male and female staphylinids of the *Mocyta fungi* species complex. Spiders were female *Dicymbium nigrum*/*brevisetosum* (1 specimen), and all immatures, which lacked developed reproductive structures.

Species were further classified as stenotopic in early successional habitats (including heathland, grassland, dunes and quarries, and fluctuating water margins), generalists present in more stable damp grassland and secondary woodland, or eurytopic. The catches from individual traps within each monitoring plot were pooled, producing single totals for each species per plot. Treatment effects were contrasted for stenotopic and generalist species. Eurytopic species with high dispersal capacities occurring more widely, such as in arable fields, were not considered further. These were most abundant in the restoration areas, for example *Oedothorax apicatus* which represented 32% of spiders captured in the study. Habitat associations of carabids were defined by Luff (2007) and Telfer (2016), staphylinids by Hyman and Parsons (1994), Lott (2009) and Lott and Anderson (2011), and spiders by Harvey *et al.* (2002).

2.2.4 Plant species composition and habitat structure

Data on plant species composition and habitat structure were collected in July 2015, in four 1 x 1 m^2 quadrats located outwards from each corner of the pitfall trap rows. Percentage cover of vascular plant species in each quadrat were recorded, with identifications from Stace (2010). Vegetation height was

measured at four random positions in each quadrat, with a custom-made drop disc of 20 cm diameter, 10 g mass. Mean vegetation height and the coefficient of variance (standard deviation divided by the mean) were calculated for each quadrat from the four height measurements. Percentage cover of moss and bare ground were recorded, and litter depth was measured to the nearest millimetre.

2.2.5 Statistical analysis

Univariate analyses were completed in the statistical software R (R Core Team, 2017). Multivariate analysis was completed with the Canoco 5.0 software package (ter Braak and Šmilauer, 2014). Precise *p*-values are reported, to allow interpretation of *p* as a measure of evidence, and to facilitate meta-analysis (Cumming and Calin-Jageman, 2017).

2.2.5.1 Species accumulation curves

To assess whether sampling effectively captured assemblage composition for each taxonomic group, rarefaction curves were constructed (Chao *et al.*, 2014), using the iNext package (Hsieh *et al.*, 2016).

2.2.5.2 Generalized linear models

Abundance of stenotopic and generalist species were compared between treatments with generalized linear models (GLMs). This approach was appropriate for count data with a non-Gaussian distribution, zero probability of negative values, and heterogeneity in the mean variance relationship (Zuur *et al.*, 2009). Pearson residuals from fitted models were calculated to check for overdispersion, and determine the appropriate distribution for the response variable (Poisson or negative binomial). These were obtained by dividing each residual by the square root of the variance (Zuur *et al.*, 2009). For model validation, to assess homogeneity, residuals were plotted versus fitted values, and versus the covariate. Cook's distance was used to assess the influence of individual observations on fitted values, with a threshold of one for further investigation (Cook and Weisberg, 1982; Fox and Weisberg, 2010). Spatial autocorrelation of residuals was checked by calculation of Moran's I in the ape package v.4.1 (Paradis *et al.*, 2004), based on a matrix of inverse Euclidean distance weights. Where this was significant ($P < 0.05$) (Table 2.2), an

autocovariate was computed using the spdep package (Bivand *et al.*, 2013; Bivand and Piras, 2015; Bivand, 2017), based on a distance-weighted function of neighbouring response values (Dormann *et al.*, 2007). This was included as an extra parameter in the model for the response variable. Differences among treatment means were examined by Tukey pair-wise comparisons. To adjust *p*-values, the family-wise error rate was controlled using the Holm-Šidák adjustment, as described in Section 2.2.5.1.

2.2.5.3 Environmental variables

GLMs were not able to consistently predict the environmental variables across all treatment classes, due to heteroscedasticity. As a result, the variables were analysed with the Kruskal–Wallis (*H*) test (Kruskal and Wallis, 1952), followed by Dunn's test to report the results among multiple pairwise comparisons (Dunn, 1964). To adjust *p*-values, the family-wise error rate was controlled using the Holm-Šidák adjustment (Holm, 1979) in the dunn.test package (Dinno, 2016). This is a progressive step-up procedure assuming dependence between tests, which are completed in order of increasing probability α/n-1. This was preferred to the classical Bonferroni correction, with tests compared to α/n, which gives a smaller or equal probability of rejecting false hypotheses for the same data (Holm, 1979).

2.2.5.4 Ordination

The relationships between species assemblages and their environment were described and visualised by ordination. This approach was chosen because of the large number of species in the analysis, and the potential to link environmental gradients to management actions (ter Braak and Verdonschot, 1995). Gradient lengths were assessed by Detrended Correspondence Analysis (DCA) of species abundance data. Axis lengths were greater than 3. This suggested the data showed unimodal response, so ordination was generated by Canonical Correspondence Analysis (CCA) (Hill and Gauch, 1980). Species abundance counts were log transformed (log (y + 1)) to reduce the effect of data being skewed by aggregations. The significance of the seven environmental variables was tested by automatic forward selection (Monte Carlo test, 500 permutations). Analysis was stopped when the remaining variables would not improve the model fit ($P < 0.05$).

2.2.5.5 Indicator species

For long-term environmental monitoring, it could be useful to assess habitat condition by the occurrence of a small number of indicator species, rather than whole invertebrate families. The association of species with treatments was therefore assessed by indicator species analysis (Dufrene and Legendre, 1997). This was completed by the indicspecies 1.7.6 package (De Cáceres and Legendre, 2009) using the multipatt command (with func = ‘‘IndVal.g’’, indicator species of individual site groups only). This gives the conditional probability of species fidelity and specificity for a treatment. These two quantities give an index value, with a permutation test (nperm = 4999) for statistical significance. The index value is group-equalised proportional to the number of sites it contains. In this study, only species with more than 10 individuals, and with an IndVal.g association index significant at alpha = 0.05 are displayed. The tests are meaningful as the site groups are management treatments, rather than classified from species (De Cáceres, 2013).

2.3 Results

2.3.1 Environmental variables

Herbaceous plant species richness differed significantly between treatments ($H_{[3,\ N=20]}$ = 47.49, P < 0.0001) (Table 2.1). This was significantly higher at reference (P_{adj} < 0.0001), phase one (P_{adj} = 0.0002), and phase two plots (P_{adj} < 0.0001) than controls, and significantly higher at reference plots than phase two plots (P_{adj} < 0.0137). Graminoid species richness differed significantly between treatments ($H_{[3,\ N=20]}$ = 19.12, P = 0.0003). This was significantly higher in reference (P_{adj} = 0.0479) and control plots (P_{adj} < 0.0001) than phase two plots. Shrub species richness differed significantly between treatments ($H_{[3,\ N=20]}$ = 48.21, P < 0.0001). This was significantly higher in control plots than phase one, phase two, or reference plots (P_{adj} < 0.0001 in all cases). The mean field layer height differed significantly between treatments ($H_{[3,\ N=20]}$ = 49.59, P < 0.0001). This was significantly higher in reference (P_{adj} = 0.0004), phase one (P_{adj} < 0.0001), and control plots (P_{adj} < 0.0001) than phase two plots, and significantly higher in control plots than reference plots (P_{adj} = 0.0139). The vegetation height coefficient of variance differed significantly between

Table 2.1 Medians (minimum-maximum) and results of Kruskal–Wallis (H) test, comparing environmental characteristics among management areas at Canvey Wick in 2015. Treatments were unmanaged controls (n=6), phase one cleared to mineral substrate in May 2014 (n=3), phase two cleared in May 2015 (n=6), and open mosaic habitat reference plots (n=5). Vegetation species were surveyed in June, and structural variables were compiled in August. For vegetation height, CoV = coefficient of variance. Post-hoc tests were Tukey pairwise comparisons after Holm-Šidák adjustment, treatments with shared superscript a-d (ranked highest to lowest) did not differ significantly ($P > 0.05$).

Variable	Control	Phase One	Phase Two	Reference	Kruskal-Wallis$_{[3, N=20]}$
Herbaceous species richness	1 (0–5) ^{c}	6 (3–12) ^{ab}	5 (0–16) ^{b}	10 (4-13) ^{a}	$H = 47.49$
Graminoid species richness	4 (2–8) ^{a}	3 (2–6) ^{ab}	3 (1–5) ^{b}	3.5 (2-7) ^{a}	$H = 19.12$
Shrub species richness	4 (1–5) ^{a}	1 (0–2) ^{b}	1 (0–2) ^{b}	0.5 (0-2) ^{b}	$H = 48.21$
Field layer height (cm)	21.0 (7.6–49.5) ^{a}	17.1 (4.4–37.5) ^{ab}	1.4 (0–26.6) ^{c}	11.6 (5.5-17.3) ^{b}	$H = 49.59$
Vegetation height CoV	0.26 (0.02–2.94) ^{b}	0.35 (0.08–5.26) ^{b}	2.44 (0–19.88) ^{a}	0.44 (0.05-2.08) ^{ab}	$H = 12.67$
Moss cover (%)	1.0 (0–5) ^{c}	5.5 (0–30) ^{a}	0.0 ± (0–0) ^{b}	5.0 (0-30) ^{a}	$H = 47.72$
Litter depth (cm)	2.5 (1.1–4.2) ^{a}	0.45 (0–2.6) ^{c}	0.0 (0–0) ^{d}	1.75 (0.9-3.1) ^{b}	$H = 63.73$
Bare ground (%)	0.5 (0–18) ^{b}	10.0 (0–38) ^{b}	85.5 (43–97) ^{a}	2.0 (0-60) ^{b}	$H = 54.80$

treatments ($H_{[3, N=20]}$ = 12.67, P = 0.0054). This was significantly higher in phase two plots than controls (P_{adj} = 0.003) or phase one plots (P_{adj} = 0.0243).

Moss cover differed significantly between treatments ($H_{[3, N=20]}$ = 47.72, P < 0.0001). This was significantly higher in reference (P_{adj} < 0.0001), phase one (P_{adj} < 0.0001), and control plots (P_{adj} = 0.0140) than phase two plots, and significantly higher in reference (P_{adj} = 0.0003) and phase one plots (P_{adj} = 0.0031) than control plots. Litter depth differed significantly between treatments ($H_{[3, N=20]}$ = 63.73, P < 0.0001). This was significantly higher in control plots than reference (P_{adj} = 0.0322), phase one (P_{adj} < 0.0001) or phase two plots (P_{adj} < 0.0001), significantly higher in reference plots than phase one (P_{adj} = 0.0191) or phase two plots (P_{adj} < 0.0001), and also significantly higher in phase one than phase two plots (P_{adj} = 0.0447). Bare ground differed significantly between treatments ($H_{[3, N=20]}$ = 54.80, P < 0.0001). This was significantly higher in phase two than phase one (P_{adj} = 0.0002), reference (P_{adj} < 0.0001), or control plots (P_{adj} < 0.0001). Although not quantified, there was surface groundwater in the restoration areas between December 2014 and May 2015, up to 15 cm deep. Control and reference plots were not flooded in this period.

2.3.2 Invertebrate summary

2.3.2.1 Carabids

2.3.2.1.1 Species richness and abundance

During the study 4,106 carabids of 80 species were identified (Appendix 9.1, Table A9.1.1, page 286). This was 89.9% of 89 species (SE ± 6) likely to be captured by pitfall traps in the sampled habitats, estimated from a rarefaction curve (Figure 2.6). In total 523 carabids were trapped at the control plots. The most abundant species were *Pterostichus strenuus* (27%), *Pterostichus melanarius* (15%) and *Notiophilus palustris* (9%). There were 17 species represented by three or fewer individuals, forming 5.4% of the catch. In total 995 carabids were trapped at the phase one restoration plots. The most abundant species were *Bembidion properans* (33%), *Bembidion quadrimaculatum* (9%) and *Stenolophus teutonus* (6%). There were 20 species represented by three or fewer individuals, forming 3.9% of the catch. In total 1,749 carabids were trapped at the phase two restoration plots. The most abundant species were *Bembidion quadrimaculatum* (67%), *Bembidion*

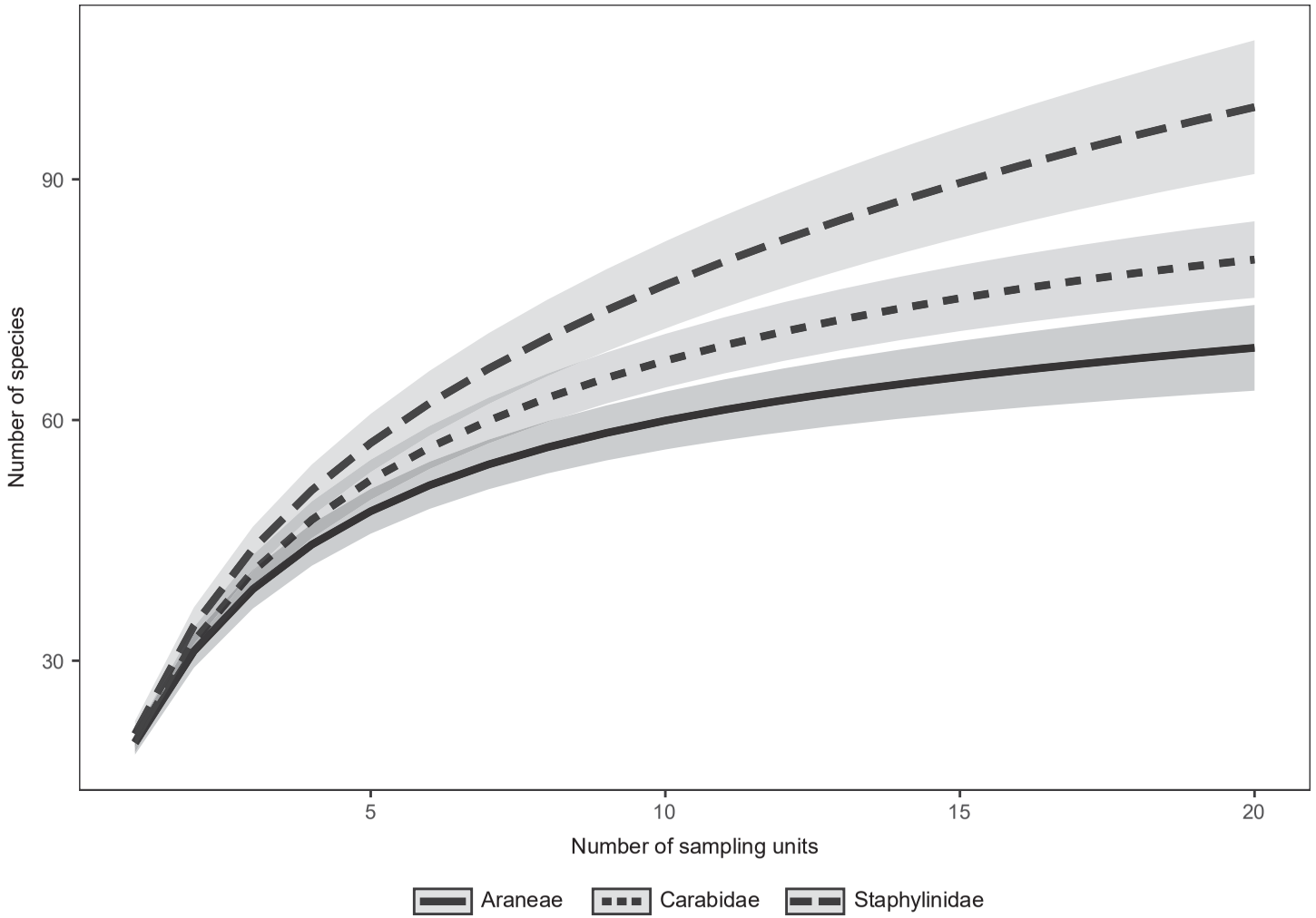

Figure 2.6 Sample-based rarefaction curves of species richness for spiders, carabids and staphylinids, based on species likely to be captured by pitfall traps in treatment, control and reference plots at Canvey Wick in 2015 (n=20). Shaded areas represent 95% confidence intervals. Curves become more shallow approaching the asymptotes of the estimated faunas.

properans (7%) and *Harpalus rufipes* (4%). There were 23 species represented by three or fewer individuals, forming 2.2% of the catch. In total 839 carabids were trapped at the mosaic habitat reference plots. The most abundant species were *Syntomus foveatus* (55%), *Amara tibialis* (21%) and *Calathus fuscipes* (6%). There were 24 species represented by three or fewer individuals, forming 4.6% of the catch.

Stenotopic carabids differed significantly in abundance ($\chi^2_{[2, N=15]} = 174.55$, $P < 0.0001$) and richness ($\chi^2_{[2, N=15]} = 29.24$, $P < 0.0001$) between treatments (Table 2.2). Abundance of stenotopic species was significantly higher in phase one and reference plots than phase two or control plots ($P_{adj} < 0.0001$ in all cases), and significantly higher in phase two than control plots ($P_{adj} = 0.0492$). Stenotopic species richness was significantly higher at phase one than control ($P_{adj} < 0.0001$), reference ($P_{adj} = 0.0322$), and phase two plots ($P_{adj} = 0.0158$), and was also significantly higher at phase two ($P_{adj} = 0.0335$) and reference plots ($P_{adj} = 0.0322$) than control plots. Generalist carabids differed significantly in abundance ($\chi^2_{[2, N=15]} = 85.24$, $P < 0.0001$) and richness ($\chi^2_{[2, N=15]} = 24.89$, $P < 0.0001$) between treatments. Abundance of generalist species was significantly higher at controls than phase two plots ($P_{adj} = 0.0391$). Abundance of generalist species was also significantly higher at phase one, phase two, and control plots, compared with reference plots ($P_{adj} < 0.0001$ in all cases). Richness of generalist species was significantly higher at phase one ($P_{adj} = 0.0014$), phase two ($P_{adj} = 0.0007$), and control plots ($P_{adj} = 0.0048$), compared with reference plots.

2.3.2.1.2 Carabid ordination

A biplot of axes one and two scores from the CCA of carabids and environmental variables (Figure 2.7) showed clear differences between the assemblages in control plots, reference plots, and both restoration phases. Axis one of the CCA explained 19% of variation in the data, separating restoration samples from reference, and control samples. Axis two explained 14% of variation in the data, and 33% in combination with axis one. This separated reference and control samples, with restoration samples in intermediate positions, although closer to reference samples overall. After Monte Carlo forward selection, two environmental variables were statistically significant in

Table 2.2 Mean ± standard error for species richness and abundance of carabids, staphylinids, and spiders at Canvey Wick, June-August 2015. Generalized linear models (χ^2, F) compare stenotopic species associated with early successional habitats, and generalists associated with grassland, woodland, or wetlands. Treatments were unmanaged controls (n=6), phase one cleared to mineral substrate in May 2014 (n=3), phase two cleared in May 2015 (n=6), and open mosaic habitat reference plots (n=5). Post-hoc tests show Tukey pairwise comparisons after Holm-Šidák adjustment, treatments with shared superscript a-c (ranked highest to lowest) did not differ significantly ($P > 0.05$). Moran's I was calculated to test for spatial correlation in GLM residuals, based on a matrix of inverse Euclidean distance weights. Where this was significant ($P < 0.05$) an autocovariate was included in the model, calculated from a distance-weighted function of neighbouring response values (Dormann *et al.*, 2007).

Invertebrate group	Variable	Control	Phase One	Phase Two	Reference	$GLM_{[3, N=20]}$	Moran's I_{GLM}
Coleoptera: Carabidae	*Stenotopic*						
	Abundance	5.5 ± 1.1 ^{c}	123.3 ± 9.5 ^{a}	12.5 ± 2.8 ^{b}	137.6 ± 37.8 ^{a}	$\chi^2 = 174.55$	0.0611
	Species richness	3.2 ± 0.7 ^{c}	13.3 ± 1.2 ^{a}	6.8 ± 0.9 ^{b}	7.0 ± 1.3 ^{b}	$F = 29.24$	0.6889
	Generalist						
	Abundance	46.5 ± 8.8 ^{a}	26.3 ± 7.8 ^{ab}	23.3 ± 4.6 ^{b}	1.2 ± 0.4 ^{c}	$\chi^2 = 85.24$	0.3255
	Species richness	5.3 ± 0.6 ^{a}	6.7 ± 1.2 ^{a}	6.5 ± 0.4 ^{a}	1.2 ± 0.4 ^{b}	$F = 24.89$	0.7535
Coleoptera: Staphylinidae	*Stenotopic*						
	Abundance	1.7 ± 0.8 ^{c}	36.3 ± 3.2 ^{a}	15.8 ± 2.2 ^{b}	1.2 ± 0.7 ^{c}	$\chi^2 = 258.87$	0.7496
	Species richness	0.7 ± 0.2 ^{b}	6.7 ± 0.7 ^{a}	6.2 ± 0.5 ^{a}	0.8 ± 0.4 ^{b}	$F = 53.26$	0.4838
	Generalist						
	Abundance	40.7 ± 12.9 ^{a}	9.3 ± 1.7 ^{b}	8.0 ± 1.6 ^{b}	5.4 ± 1.4 ^{b}	$\chi^2 = 46.70$	0.7706
	Species richness	9.8 ± 0.9 ^{a}	5.3 ± 0.9 ^{b}	4.2 ± 0.3 ^{b}	1.8 ± 0.7 ^{c}	$F = 91.67$	0.0058
Araneae	*Stenotopic*						
	Abundance	7.7 ± 1.9 ^{c}	71.0 ± 13.1 ^{a}	31.3 ± 3.7 ^{b}	24.4 ± 4.5 ^{b}	$\chi^2 = 82.55$	0.6553
	Species richness	2.5 ± 0.6 ^{b}	6.3 ± 2.0 ^{a}	4.0 ± 0.7 ^{ab}	7.4 ± 0.8 ^{a}	$F = 16.23$	0.0913
	Generalist						
	Abundance	36.3 ± 6.3 ^{a}	10.7 ± 1.9 ^{b}	6.2 ± 0.5 ^{b}	11.0 ± 4.2 ^{b}	$\chi^2 = 54.19$	0.1749
	Species richness	7.8 ± 1.0 ^{a}	5.7 ± 0.7 ^{ab}	3.0 ± 0.7 ^{b}	3.8 ± 0.5 ^{b}	$F = 15.62$	0.6452

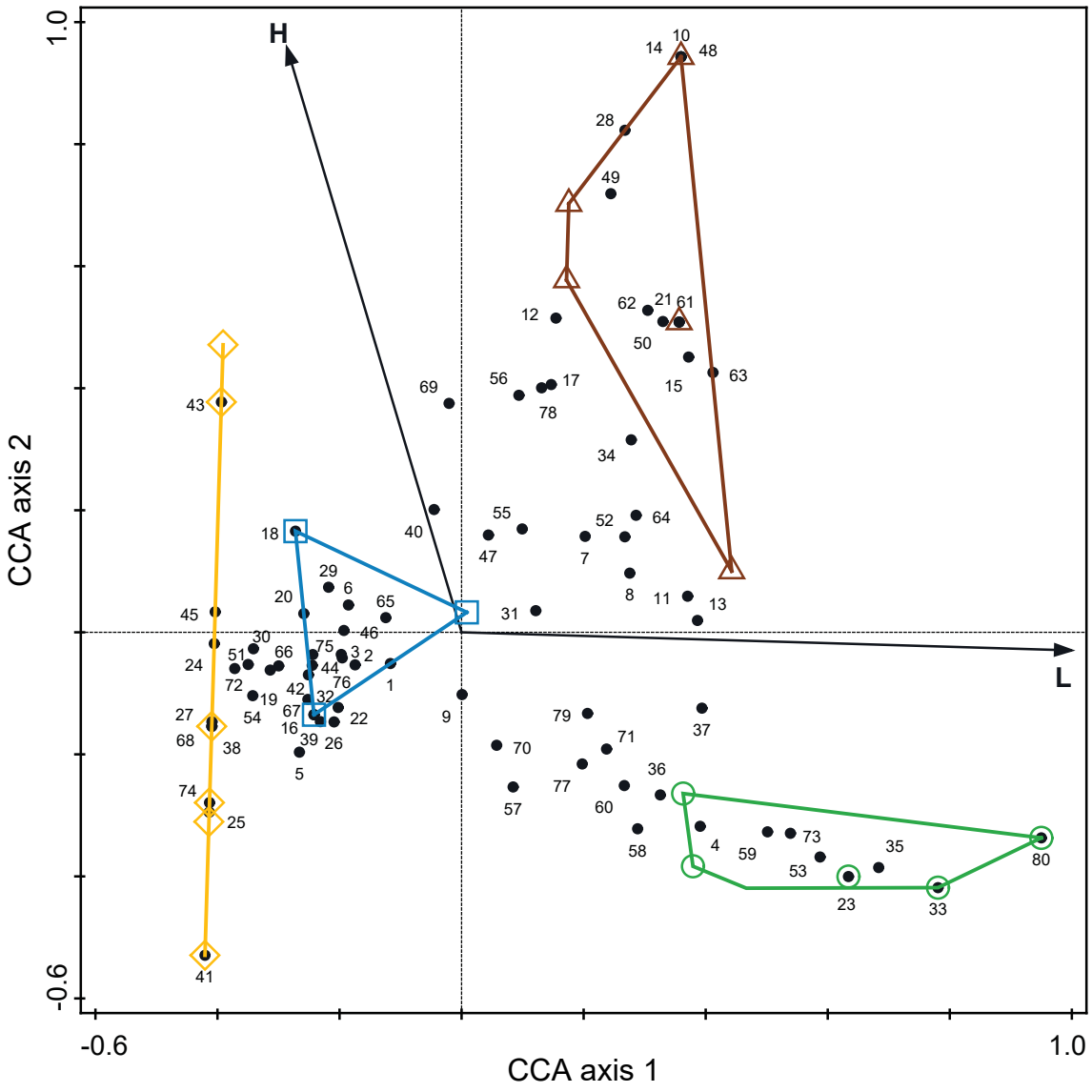

Figure 2.7 Biplot scores for carabid species (•), environmental variables (→), and treatment plots, calculated with Canonical Correspondence Analysis of the carabid assemblage at Canvey Wick in 2015, as determined from captures in pitfall traps. Management treatments are coded: (◯) unmanaged control; (□) phase one restoration cleared to mineral substrate in May 2014; (◇) phase two cleared in May 2015; (△) open mosaic habitat reference plot. Environmental variables are L, litter depth (P_{adj} = 0.016), and H, herbaceous plant species richness (P_{adj} = 0.016). Numerical references for species names: 1 = *Acupalpus brunnipes*, 2 = *Acupalpus dubius*, 3 = *Acupalpus parvulus*, 4 = *Agonum emarginatum*, 5 = *Agonum marginatum*, 6 = *Agonum muelleri*, 7 = *Amara aenea*, 8 = *Amara communis*, 9 = *Amara convexior*, 10 = *Amara curta*, 11 = *Amara eurynota*, 12 = *Amara familiaris*, 13 = *Amara lunicollis*, 14 = *Amara montivaga*, 15 = *Amara ovata*, 16 = *Amara similata*, 17 = *Amara tibialis*, 18 = *Anchomenus dorsalis*, 19 = *Anisodactylus binotatus*, 20 = *Asaphidion flavipes*, 21 = *Badister bullatus*, 22 = *Bembidion assimile*, 23 = *Bembidion biguttatum*, 24 = *Bembidion femoratum*, 25 = *Bembidion illigeri*, 26 = *Bembidion lunulatum*, 27 = *Bembidion minimum*, 28 = *Bembidion obtusum*, 29 = *Bembidion properans*, 30 = *Bembidion quadrimaculatum*, 31 = *Brachinus crepitans*, 32 = *Bradycellus verbasci*, 33 = *Calathus cinctus*, 34 = *Calathus fuscipes*, 35 = *Calathus rotundicollis*, 36 = *Carabus nemoralis*, 37 = *Carabus violaceus*, 38 = *Chlaenius nigricornis*, 39 = *Clivina fossor*, 40 = *Curtonotus aulicus*, 41 = *Curtonotus convexiusculus*, 42 = *Dyschirius aeneus*, 43 = *Dyschirius luedersi*, 44 = *Dyschirius politus*, 45 = *Elaphropus parvulus*, 46 = *Elaphrus riparius*, 47 = *Harpalus affinis*, 48 = *Harpalus anxius*, 49 = *Harpalus attenuatus*, 50 = *Harpalus rubripes*, 51 = *Harpalus rufipes*, 52 = *Harpalus tardus*, 53 = *Leistus ferrugineus*, 54 = *Loricera pilicornis*, 55 = *Microlestes maurus*, 56 = *Microlestes minutulus*, 57 = *Nebria brevicollis*, 58 = *Notiophilus biguttatus*, 59 = *Notiophilus palustris*, 60 = *Notiophilus substriatus*, 61 = *Ophonus azureus*, 62 = *Ophonus puncticeps*, 63 = *Ophonus rufibarbis*, 64 = *Panagaeus bipustulatus*, 65 = *Paradromius linearis*, 66 = *Poecilus cupreus*, 67 = *Pterostichus gracilis*, 68 = *Pterostichus macer*, 69 = *Pterostichus madidus*, 70 = *Pterostichus melanarius*, 71 = *Pterostichus niger*, 72 = *Pterostichus nigrita*, 73 = *Pterostichus strenuus*, 74 = *Pterostichus vernalis*, 75 = *Stenolophus mixtus*, 76 = *Stenolophus teutonus*, 77 = *Stomis pumicatus*, 78 = *Syntomus foveatus*, 79 = *Syntomus truncatellus*, 80 = *Trechus quadristriatus*.

structuring the ordination. Litter depth was strongly positively correlated with axis one (F-ratio = 4.3, P_{adj} = 0.016). Forb species richness was positively correlated with axis two (F-ratio = 3.4, P_{adj} = 0.016). The other environmental variables measured either correlated with these, or did not describe a significant proportion of the variation in the carabid community.

Stenotopic species typical of recently scraped plots formed a limited but distinct group in the ordination, particularly *Bembidion femoratum*, *Bembidion illigeri* and *Elaphropus parvulus*. Another group of species was associated with plots one year after disturbance, including *Asaphidion flavipes*, *Dyschirius aeneus*, *Dyschirius politus*, *Elaphrus riparius* and *Stenolophus teutonus*. A subset of graminivorous carabids was found at reference plots, including *Amara montivaga*, *Amara curta*, *Harpalus attenuatus* and *Harpalus rubripes*. *Bembidion lunulatum* and *Poecilus cupreus* occurred in similar numbers on plots immediately after scraping, and one year later. *Amara familiaris*, *Amara tibialis* and *Syntomus foveatus* were most frequent at phase one and reference plots. No species were particularly frequent at control plots, although *Amara convexior*, *Amara aenea* and *Amara eurynota* occurred in all treatments.

2.3.2.2 Staphylinids

2.3.2.2.1 Species richness and abundance

During the study 1,925 staphylinids of 99 species were identified (Appendix 9.1, Table A9.1.2, page 286). This was 74.4% of 133 species (SE ± 3) likely to be captured by pitfall traps in the sampled habitats, estimated from a rarefaction curve (Figure 2.6). In total 728 staphylinids were trapped at the control plots. The most abundant species were *Drusilla canaliculata* (33%), *Tachyporus nitidulus* (8%) and *Philonthus succicola* (7%). There were 30 species represented by three or fewer individuals, forming 6.9% of the catch. In total 312 staphylinids were trapped at the phase one restoration plots. The most abundant species were *Aleochara brevipennis* (25%), *Bledius gallicus* (20%) and *Carpelimus incongruus* (4%). There were 30 species represented by three or fewer individuals, forming 13.1% of the catch. In total 788 staphylinids were trapped at the phase two restoration plots. The most abundant species were *Anotylus rugosus* (35%), *Aleochara bipustulata* (13%) and *Aleochara brevipennis* (12%). There were 16 species represented by three or fewer

individuals, forming 3.0% of the catch. In total 97 individuals were trapped at the mosaic habitat reference plots. The most abundant species were *Platydracus stercorarius* (25%), *Pella limbata* (14%) and *Drusilla canaliculata* (11%). There were 17 species represented by three or fewer individuals, forming 24.7% of the catch.

Stenotopic staphylinids differed significantly in abundance ($\chi^2_{[3, N=20]} = 258.87$, $P < 0.0001$) and richness ($F_{[3, N=20]} = 53.26$, $P < 0.0001$) between treatments (Table 2.2). Abundance of stenotopic species was significantly higher in phase one and phase two plots than reference or control plots ($P_{adj} < 0.0001$ in all cases), and was also significantly higher in phase one plots than phase two plots ($P_{adj} < 0.0001$). Stenotopic species richness was significantly higher in phase one and phase two plots than control or reference plots ($P_{adj} < 0.0001$ in all cases). Generalist staphylinids differed significantly in abundance ($\chi^2_{[3, N=20]} = 46.70$, $P < 0.0001$) and richness ($\chi^2_{[3, N=20]} = 15.22$, $P = 0.0004$) between treatments. Abundance of generalist species was significantly higher in control plots than phase one, phase two, or reference plots ($P_{adj} < 0.0001$ in all cases), and also significantly higher in phase one ($P_{adj} < 0.0001$) and phase two plots ($P_{adj} = 0.0003$) than reference plots. Richness of generalist species was significantly higher in control plots than reference ($P_{adj} < 0.0001$) or phase two plots ($P_{adj} = 0.0019$).

2.3.2.2.2 Staphylinid ordination

A biplot of axes one and two scores from the CCA of staphylinids and environmental variables (Figure 2.8) showed differences between the assemblages in both restoration phases, and partially overlapping reference and control plots. Axis one of the CCA explained 18% of variation in the data, separating phase two, and to a lesser extent phase one samples, from control and reference plots. Axis two explained 10% of variation in the data, and 28% in combination with axis one. This separated phase one plots from phase two, reference, and control plots. After Monte Carlo forward selection, three environmental variables were statistically significant in structuring the ordination. Larger areas of bare ground were negatively correlated with axis one (F-ratio = 2.7, $P_{adj} = 0.016$). Deeper litter was positively correlated with axis one (F-ratio = 2.4, $P_{adj} = 0.016$). Taller vegetation was positively correlated with axes

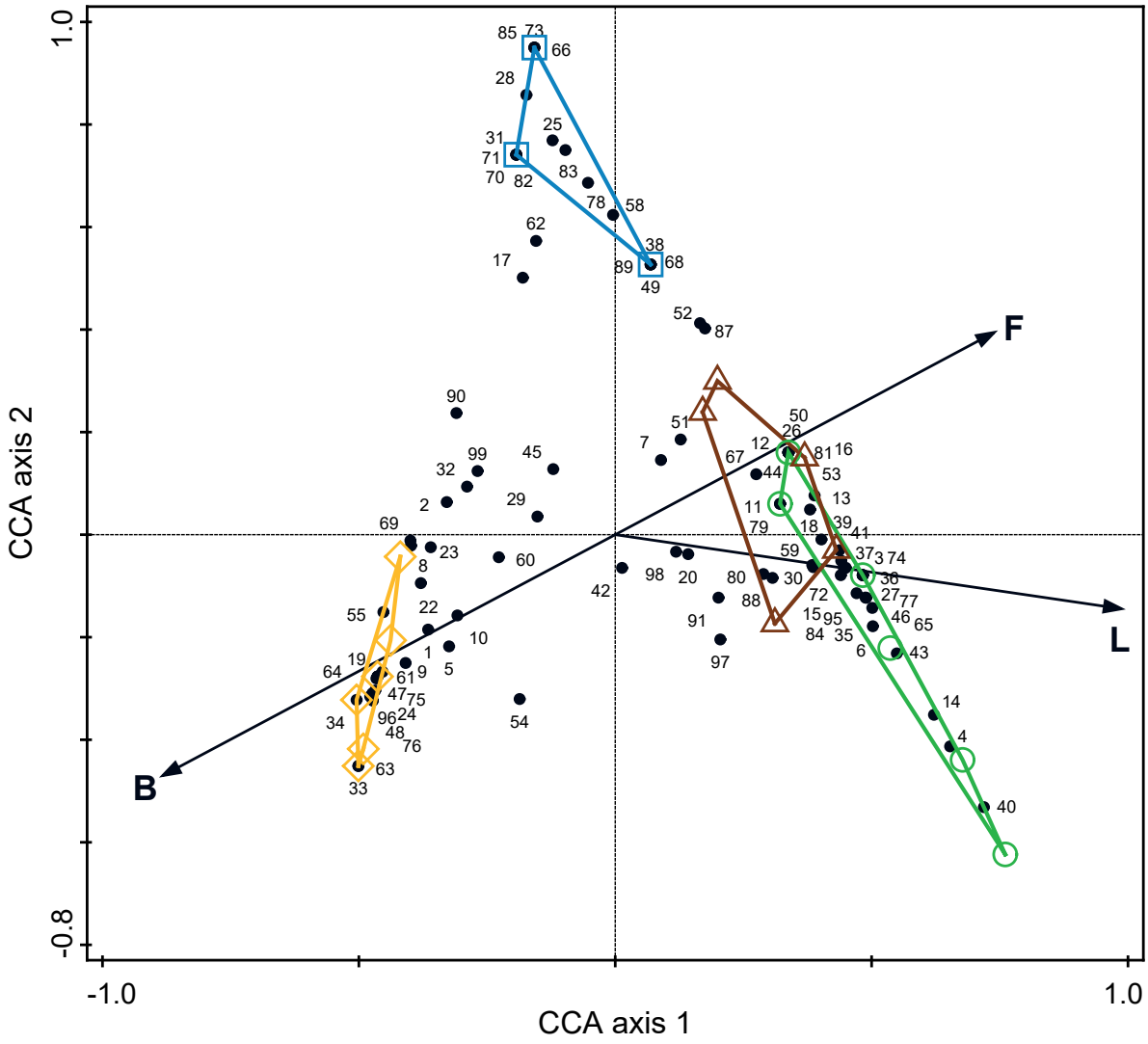

Figure 2.8 Biplot scores for staphylinid species (●), environmental variables (→), and treatment plots, calculated with Canonical Correspondence Analysis of the staphylinid assemblage at Canvey Wick in 2015, as determined from captures in pitfall traps. Management treatments are coded (○) unmanaged control; (□) phase one restoration cleared to mineral substrate in May 2014; (◇) phase two cleared in May 2015; (△) open mosaic habitat reference plot. Environmental variables are L, litter depth (P_{adj} = 0.016); B, bare ground (P_{adj} = 0.016); F, field layer height (P_{adj} = 0.05). Numerical references for species names: 1 = *Aleochara bipustulata*, 2 = *Aleochara brevipennis*, 3 = *Aleochara curtula*, 4 = *Aleochara sparsa*, 5 = *Aloconota gregaria*, 6 = *Amarochara forticornis*, 7 = *Amischa analis*, 8 = *Amischa decipiens*, 9 = *Anotylus nitidulus*, 10 = *Anotylus rugosus*, 11 = *Anotylus sculpturatus*, 12 = *Anotylus tetracarinatus*, 13 = *Atheta aquatica*, 14 = *Atheta castanoptera*, 15 = *Atheta vaga*, 16 = *Autalia rivularis*, 17 = *Bledius gallicus*, 18 = *Brachygluta fossulata*, 19 = *Brundinia marina*, 20 = *Callicerus rigidicornis*, 21 = *Calodera protensa*, 22 = *Carpelimus bilineatus*, 23 = *Carpelimus corticinus*, 24 = *Carpelimus gracilis*, 25 = *Carpelimus incongruus*, 26 = *Chaetida longicornis*, 27 = *Cordalia obscura*, 28 = *Cypha longicornis*, 29 = *Dinaraea angustula*, 30 = *Drusilla canaliculata*, 31 = *Gabrius appendiculatus*, 32 = *Gabrius breviventer*, 33 = *Gabrius nigritulus*, 34 = *Gyrohypnus angustatus*, 35 = *Gyrohypnus wagneri*, 36 = *Gyrophaena bihamata*, 37 = *Gyrophaena fasciata*, 38 = *Ilyobates nigricollis*, 39 = *Ilyobates propinquus*, 40 = *Ischnosoma splendidum*, 41 = *Lathrobium brunnipes*, 42 = *Lathrobium geminum*, 43 = *Liogluta longiuscula*, 44 = *Liogluta pagana*, 45 = *Meotica exilis*, 46 = *Microdota indubia*, 47 = *Nehemitropia lividipennis*, 48 = *Neobisnius procerulus*, 49 = *Neobisnius villosulus*, 50 = *Ocypus fuscatus*, 51 = *Ocypus olens*, 52 = *Oligota pumilio*, 53 = *Omalium septentrionis*, 54 = *Othius punctulatus*, 55 = *Othius subuliformis*, 56 = *Oxypoda brachyptera*, 57 = *Oxypoda brevicornis*, 58 = *Paederus littoralis*, 59 = *Pella limbata*, 60 = *Philhygra elongatula*, 61 = *Philhygra palustris*, 62 = *Philonthus cognatus*, 63 = *Philonthus fumarius*, 64 = *Philonthus micans*, 65 = *Philonthus succicola*, 66 = *Planeustomus palpalis*, 67 = *Platydracus stercorarius*, 68 = *Platystethus alutaceus*, 69 = *Platystethus nitens*, 70 = *Pselaphus heisei*, 71 = *Quedius boops*, 72 = *Quedius curtipennis*, 73 = *Quedius levicollis*, 74 = *Quedius picipes*, 75 = *Rugilus erichsonii*, 76 = *Rugilus orbiculatus*, 77 = *Rugilus rufipes*, 78 = *Staphylinus dimidiaticornis*, 79 = *Stenichnus bicolor*, 80 = *Stenichnus scutellaris*, 81 = *Stenus bimaculatus*, 82 = *Stenus brunnipes*, 83 = *Stenus canaliculatus*, 84 = *Stenus clavicornis*, 85 = *Stenus juno*, 86 = *Stenus nanus*, 87 = *Stenus pusillus*, 88 = *Tachinus rufipes*, 89 = *Tachyporus dispar*, 90 = *Tachyporus hypnorum*, 91 = *Tachyporus nitidulus*, 92 = *Tachyporus pusillus*, 93 = *Tachyporus scitulus*, 94 = *Tasgius ater*, 95 = *Tasgius morsitans*, 96 = *Thecturota marchii*, 97 = *Xantholinus elegans*, 98 = *Xantholinus linearis*, 99 = *Xantholinus longiventris*.

one and two (F-ratio = 1.9, P_{adj} = 0.05). Other environmental variables either correlated with these, or did not describe a significant proportion of the variation in the staphylinid community.

Most stenotopic species were particularly associated with restored plots. The staphylinids *Neobisnius procerulus*, *Thecturota marchii*, *Rugilus orbiculatus*, *Philhygra palustris* and *Carpelimus gracilis* correlated with increased areas of bare ground associated with the most recently scraped plots. There was also a cluster of staphylinids associated with plots a year after scraping, including *Bledius gallicus*, *Planeustomus palpalis*, *Stenus canaliculatus* and *Carpelimus incongruus*. *Xantholinus elegans* correlated with unmanaged plots and reference plots that had increased litter depth.

2.3.2.3 Spiders

2.3.2.3.1 Species richness and abundance

During the study 2,694 spiders of 69 species were identified (Appendix 9.1, Table A9.1.3, page 286). This was 84.1% of 82 species (SE ± 10) likely to be captured by pitfall traps in the sampled habitats, estimated from a rarefaction curve (Figure 2.6). In total 476 spiders were trapped at the control plots. The most abundant species were *Agraecina striata* (19%), *Tenuiphantes tenuis* (9%) and *Trochosa terricola* (9%). There were 20 species represented by three or fewer individuals, forming 5.9% of the catch. In total 762 spiders were trapped at the phase one restoration plots. The most abundant species were *Oedothorax apicatus* (24%), *Erigone dentipalpis* (16%) and *Pardosa agrestis* (15%). There were 14 species represented by three or fewer individuals, forming 2.3% of the catch. In total 1,108 spiders were trapped at the phase two restoration plots. The most abundant species were *Oedothorax apicatus* (52%), *Erigone dentipalpis* (13%) and *Pardosa agrestis* (13%). There were 25 species represented by three or fewer individuals, forming 2.0% of the catch. In total 258 spiders were trapped in the mosaic habitat reference plots. The most abundant species were *Zodarion italicum* (16%), *Alopecosa pulverulenta* (12%) and *Drassyllus pusillus* (11%). There were 16 species represented by three or fewer individuals, forming 10.9% of the catch.

Stenotopic spiders differed significantly in abundance ($\chi^2_{[3, N=20]}$ = 82.55, P < 0.0001) and richness ($\chi^2_{[3, N=20]}$ = 16.23, P = 0.0261) between treatments (Table

2.2). Abundance of stenotopic species was significantly higher in phase one, phase two, and reference plots than control plots ($P_{adj} < 0.0001$ in all cases), and also significantly higher in phase one plots than phase two ($P_{adj} < 0.0019$) or reference plots ($P_{adj} < 0.0001$). Richness of stenotopic species was significantly higher in phase one ($P_{adj} = 0.0427$) and reference plots ($P_{adj} = 0.0024$) than control plots. Generalist spiders differed significantly in abundance ($\chi^2_{[3, N=20]} = 54.19$, $P < 0.0001$) and richness ($\chi^2_{[3, N=20]} = 15.63$, $P = 0.0014$) between treatments. Abundance of generalist species was significantly higher in control plots than phase one, phase two, or reference plots ($P_{adj} < 0.0001$ in all cases). Richness of generalist species was significantly higher in control plots than phase two ($P_{adj} = 0.0032$) or reference plots ($P_{adj} = 0.0468$).

2.3.2.3.2 Spider ordination

A biplot of axes one and two scores from the CCA of spiders and environmental variables (Figure 2.9) showed differences between the assemblages in the restoration areas, and reference and control plots. 24% of variation in the data was explained by axis one of the CCA ordination, which separated restored plots from control plots. Axis two explained 23% of variation in the spider data, and 47% in combination with axis one. This separated reference plots from restored plots, and control plots. After Monte Carlo forward selection, one environmental variable was statistically significant in structuring the ordination. Litter depth was positively correlated with axis one (F-ratio = 5.8, $P_{adj} = 0.016$). Other environmental variables either correlated with this, or did not describe a significant proportion of the variation in the spider community. *Erigone arctica* was the only stenotopic species limited to newly scraped plots. *Arctosa leopardus* and *Pardosa agrestis* were most abundant in both restoration areas. *Ozyptila simplex* mostly occurred on undisturbed plots and on plots one year after scraping. The largest group of stenotopic species was particularly associated with reference plots, including *Argenna subnigra*, *Drassodes cupreus*, *Drassyllus pusillus*, *Haplodrassus signifer*, *Trachyzelotes pedestris*, *Zelotes latreillei* and *Zodarion italicum*.

2.2.3.3 Indicator species

Indicator species were identified for all three invertebrate groups, although there were no spider indicators for the phase two restoration (Table 2.3). For

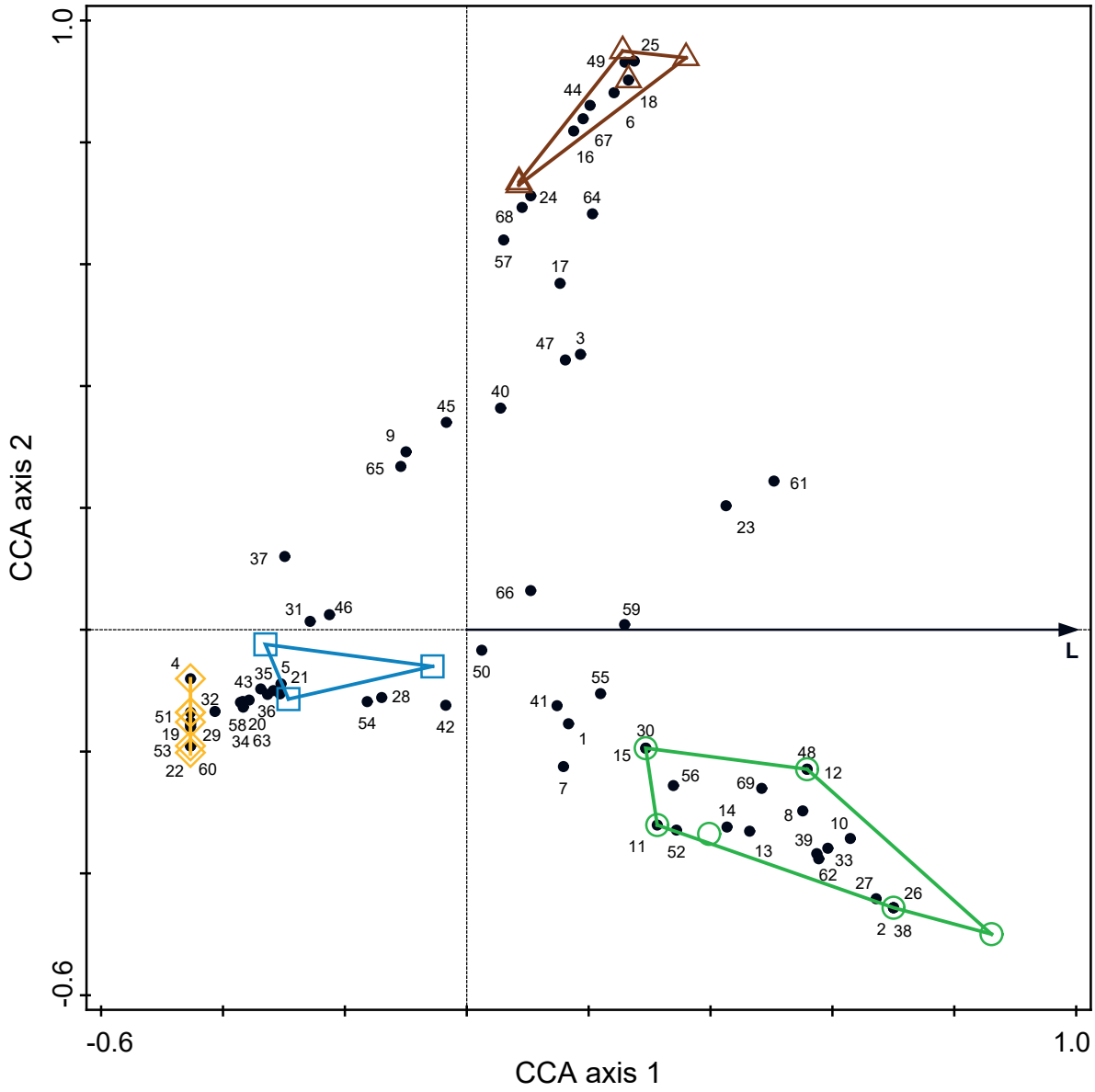

Figure 2.9 Biplot scores for spider species (●), environmental variables (→), and treatment plots, calculated with Canonical Correspondence Analysis of the spider assemblage at Canvey Wick in 2015, as determined from captures in pitfall traps. Management treatments are coded (◯) unmanaged control; (□) phase one restoration cleared to mineral substrate in May 2014; (◇) phase two cleared in May 2015; (△) open mosaic habitat reference plot. The environmental variable is L, litter depth (P_{adj} = 0.016). Numerical references for species names: 1 = *Agraecina striata*, 2 = *Agroeca inopina*, 3 = *Alopecosa pulverulenta*, 4 = *Antistea elegans*, 5 = *Arctosa leopardus*, 6 = *Argenna subnigra*, 7 = *Bathyphantes gracilis*, 8 = *Bathyphantes parvulus*, 9 = *Cheiracanthium virescens*, 10 = *Clubiona comta*, 11 = *Clubiona pallidula*, 12 = *Clubiona reclusa*, 13 = *Diplocephalus picinus*, 14 = *Diplostyla concolor*, 15 = *Dismodicus bifrons*, 16 = *Drassodes cupreus*, 17 = *Drassyllus pusillus*, 18 = *Dysdera crocata*, 19 = *Erigone arctica*, 20 = *Erigone atra*, 21 = *Erigone dentipalpis*, 22 = *Euophrys frontalis*, 23 = *Hahnia nava*, 24 = *Haplodrassus signifer*, 25 = *Heliophanus flavipes*, 26 = *Linyphia hortensis*, 27 = *Maso sundevalli*, 28 = *Meioneta beata*, 29 = *Meioneta rurestris*, 30 = *Meioneta saxatilis*, 31 = *Micaria pulicaria*, 32 = *Micrargus subaequalis*, 33 = *Microneta viaria*, 34 = *Oedothorax apicatus*, 35 = *Oedothorax fuscus*, 36 = *Oedothorax retusus*, 37 = *Ostearius melanopygius*, 38 = *Ozyptila brevipes*, 39 = *Ozyptila praticola*, 40 = *Ozyptila sanctuaria*, 41 = *Ozyptila simplex*, 42 = *Pachygnatha degeeri*, 43 = *Pardosa agrestis*, 44 = *Pardosa nigriceps*, 45 = *Pardosa palustris*, 46 = *Pardosa prativaga*, 47 = *Pardosa pullata*, 48 = *Pardosa saltans*, 49 = *Pelecopsis parallela*, 50 = *Phrurolithus festivus*, 51 = *Pirata piraticus*, 52 = *Pocadicnemis juncea*, 53 = *Porrhomma microphthalmum*, 54 = *Talavera aequipes*, 55 = *Tenuiphantes tenuis*, 56 = *Tiso vagans*, 57 = *Trachyzelotes pedestris*, 58 = *Trochosa ruricola*, 59 = *Trochosa terricola*, 60 = *Walckenaeria acuminata*, 61 = *Walckenaeria antica*, 62 = *Walckenaeria atrotibialis*, 63 = *Walckenaeria vigilax*, 64 = *Xysticus cristatus*, 65 = *Xysticus kochi*, 66 = *Zelotes apricorum*, 67 = *Zelotes latreillei*, 68 = *Zodarion italicum*, 69 = *Zora spinimana*.

Table 2.3 List of significant indicator species ($p < 0.05$, Monte Carlo test, 4,999 permutations) for carabids, staphylinids, and spiders in management areas at Canvey Wick in 2015, based on occurrence of more than 10 individuals in separate treatments. Fidelity describes limitation in species occurrence among treatments (1=single). Constancy describes frequency of occurrence within a treatment (1=constant). Plus symbol ($^+$) by species name denotes status as a stenotopic indicator of early successional habitat in this study.

Invertebrate group	Treatment	Taxon	Individuals	Fidelity	Constancy	Index value	Significance
Coleoptera: Carabidae	Control	*Leistus ferrugineus*	16	1.0000	1.0000	1.000	0.001
		Pterostichus strenuus	140	0.9929	1.0000	0.996	0.001
		Notiophilus palustris	48	0.9388	1.0000	0.969	0.001
		Notiophilus biguttatus	49	0.8627	1.0000	0.929	0.009
	Phase One	*Elaphrus riparius*$^+$	56	1.0000	1.0000	1.000	0.001
		Stenolophus teutonus$^+$	66	0.9846	1.0000	0.992	0.001
		Acupalpus parvulus	36	0.9714	1.0000	0.986	0.001
		Dyschirius politus$^+$	52	0.9388	1.0000	0.969	0.002
		Stenolophus mixtus	22	0.9268	1.0000	0.963	0.001
		Amara convexior$^+$	56	0.7062	1.0000	0.840	0.025
		Asaphidion flavipes$^+$	13	1.0000	0.6667	0.816	0.026
	Phase Two	*Bembidion minimum*	35	1.0000	1.0000	1.000	0.001
		Pterostichus nigrita	51	0.9245	1.0000	0.962	0.001
		Harpalus rufipes	75	0.9091	1.0000	0.953	0.002
		Bembidion femoratum$^+$	16	1.0000	0.8333	0.913	0.003
		Clivina fossor	18	0.8333	0.6667	0.745	0.028
	Reference	*Calathus fuscipes*	59	0.8864	1.0000	0.941	0.001
		Harpalus rubripes$^+$	10	1.0000	0.8000	0.894	0.003
Coleoptera: Staphylinidae	Control	*Quedius curtipennis*	25	0.9231	1.0000	0.961	0.001
		Stenus clavicornis	22	0.8120	1.0000	0.901	0.004
		Philonthus succicola	52	0.9623	0.8333	0.895	0.009
		Tachinus rufipes	46	0.9348	0.8333	0.883	0.006
		Tasgius morsitans	18	0.8333	0.8333	0.833	0.010
		Quedius picipes	13	0.7746	0.8333	0.803	0.016

Invertebrate group	Treatment	Taxon	Individuals	Fidelity	Constancy	Index value	Significance
	Phase One	*Carpelimus incongruus*$^{+}$	14	1.0000	1.0000	1.000	0.003
		Stenus canaliculatus^{+}	10	1.0000	1.0000	1.000	0.003
		Paederus littoralis^{+}	14	0.9559	1.0000	0.978	0.003
		Bledius gallicus^{+}	68	0.9618	1.0000	0.981	0.002
	Phase Two	*Neobisnius procerulus*$^{+}$	29	1.0000	1.0000	1.000	0.001
		Anotylus nitidulus	22	0.9545	1.0000	0.977	0.001
		Anotylus rugosus	284	0.9405	1.0000	0.970	0.004
		Aleochara bipustulata	109	0.9083	1.0000	0.953	0.002
		Aloconota gregaria	35	0.8611	1.0000	0.928	0.002
	Reference	*Platydracus stercorarius*	29	0.8726	1.0000	0.910	0.002
Araneae	Control	*Diplocephalus picinus*	26	0.9615	1.0000	0.981	0.001
		Diplostyla concolor	13	0.9231	1.0000	0.961	0.001
		Microneta viaria	10	1.0000	0.8333	0.913	0.003
		Ozyptila praticola	11	1.0000	0.8333	0.913	0.003
		Walckenaeria atrotibialis	27	1.0000	0.8333	0.913	0.002
		Zora spinimana	30	0.9295	0.8833	0.880	0.007
		Bathyphantes parvulus	24	0.9524	0.6667	0.797	0.020
	Phase One	*Pardosa prativaga*	10	0.8235	1.0000	0.907	0.011
		Pachygnatha degeeri	28	0.8163	1.0000	0.904	0.010
	Reference	*Alopecosa pulverulenta*	41	0.7669	1.0000	0.875	0.007
		Zodarion italicum^{+}	43	0.9425	0.8000	0.868	0.009
		Trachyzelotes pedestris^{+}	17	0.8571	0.8000	0.828	0.013
		Pelecopsis parallela	14	1.0000	0.6000	0.775	0.022

carabids, *Leistus ferrugineus* was a constant at control plots, and not recorded in the restoration areas. *Pterostichus strenuus*, *Notiophilus palustris* and *Notiophilus biguttatus* were also constants but occurred in other treatments. Among stenotopic species, *Elaphrus riparius* was constant in the phase one plots and not found in the other treatments. *Stenolophus teutonus*, *Amara convexior* and *Dyschirius politus* were also constant at phase one plots but not with complete fidelity. *Asaphidion flavipes*, *Bembidion femoratum* and *Harpalus rubripes* showed complete fidelity to phase one, phase two, and reference plots respectively, but were not constants.

No staphylinids were specific to control or reference plots, although *Quedius curtipennis* and *Stenus clavicornis* were constants at control plots. Among stenotopic species, *Carpelimus incongruus* and *Stenus canaliculatus* were constant and restricted to phase one. *Paederus littoralis* and *Bledius gallicus* were also constant in phase one, but occurred in other treatments. *Neobisnius procerulus* was constant and restricted to phase two. For spiders *Diplocephalus picinus* and *Diplostyla concolor* were specific to control plots although not constant. *Microneta varia*, *Ozyptila praticola*, *Walckenaeria atrotibialis* were constant in the control plots but also occurred in other treatments. No stenotopic spiders were identified as indicators in phase one or phase two. *Pardosa prativaga* and *Pachygnatha degeeri* were both constants in phase one but also occurred in other treatments. *Zodarion italicum* and *Trachyzelotes pedestris* were stenotopic indicators at reference plots, although without complete fidelity or constancy.

2.4 Discussion

2.4.1 Restoring early successional habitat

This study focused on the effects of resetting vegetation succession, in an area of secondary woodland previously used for disposal of sandy river dredgings. Habitat restoration benefitted stenotopic early successional carabids, staphylinids, and spiders. Responses were similar for each group, with greater abundance in disturbed areas than controls, and the greatest abundance a year after disturbance. There were distinct groups of species immediately after scraping, and one year later. These were also separated from open mosaic habitat reference plots. Abundance of generalist species decreased after

scraping for carabids, staphylinids and spiders. Abundance of carabids had recovered one year later, when the abundance of staphylinids and spiders was still significantly lower.

2.4.2 Response of carabids

Abundance of stenotopic carabids was significantly greater in restored areas, compared with unmanaged plots. Shrub and tree encroachment had a similar effect in dry sandy grasslands, where xerophilic habitat specialists were replaced by woodland and eurytopic carabids (Buchholz *et al.*, 2013). Although clearances benefited open habitat specialists, it should be noted that thirty three disturbance associated carabids were recorded in control plots in the present study (73% in June). Spring breeding species might need patches of scrub at other times, as wetland carabids were found overwintering in meadows and woodland within 50 metres of fens (Andersen, 2011). Control plots also supported significantly greater abundance of woodland and grassland carabids, compared with phase two or reference plots. This was potentially linked to deeper leaf litter, including prey availability, increased humidity, and amelioration in microclimate (Magura *et al.*, 2005; Schirmel and Buchholz, 2011). One year after scraping, generalist abundance had recovered, possibly because the closed canopy woodland was less than 30 years old, and contained many edge associated species (Magura *et al.*, 2001).

Abundance and richness of specialist carabids did not increase immediately after scraping in sandy grassland beside forestry trackways (Pedley *et al.*, 2013). The treatment plots in the present study were 1 ha and in an open, coastal location. The forestry plots were 0.0675 ha (Pedley *et al.*, 2013) and plantations are probably less permeable for dispersing invertebrates, as demonstrated for butterflies (Ricketts, 2001). There could also be differences in the regional pool of potential colonists, as carabids were abundant in the earliest stages of succession after dune slack restoration (Desender *et al.*, 2007). While newly scraped areas were valuable for carabids, stenotopic species were six times more abundant a year later. Increased time since disturbance was also the most significant factor explaining greater numbers of specialist carabids on urban brownfields (Small *et al.*, 2006). Two months after scraping in the present study, plots were extensively bare (81.8% ± SE 2.6).

Many carabids respond to vegetation structure (Brose, 2003a), and there were probably also insufficient resources for graminivorous species.

There was only limited convergence between restoration and reference assemblages. Although overall abundance was lower at newly scraped plots, six stenotopic species were not recorded elsewhere. Reference plots were characterised by phytophagous carabids, including differentiation from seven species in the genera *Amara*, *Harpalus* and *Ophonus*. This was possibly linked to increased frequency of leguminous plants (Talarico *et al.*, 2016), particularly black medick *Medicago lupulina*. There were also differences in habitat structure at restored plots. Scraping removed 15-20 cm of substrate, creating areas with above-surface groundwater in the winter. Wet, partly vegetated sand was absent from the rest of the site, which suggests that local habitat was not the source of all colonising carabids. Stenotopic species in the restored areas have consistently been associated with hydric environments in other research. *Bembidion femoratum* was an indicator of recently scraped plots, and has been recorded at ephemeral pools in arable land (Brose, 2003b). *Dyschirius politus*, *Asaphidion flavipes* and *Elaphrus riparius* were indicators one year after scraping, and also colonised reprofiled river banks (Günther and Assmann, 2005).

2.4.3 Response of staphylinids

The response of staphylinids to habitat restoration was partly similar to carabids, with significantly greater abundance of stenotopic species in restored areas. This is consistent with the importance of semi-natural early successional environments for staphylinids, including exposed riverine sediments (Eyre *et al.*, 2001). Although there was no significant difference in richness of stenotopic species between the restored areas, these were separated by community composition. *Neobisnius procerulus* was only recorded in the phase two restoration, and has mainly been reported from secondary habitats with soft sediments in the UK, including gravel pits (Denton, 2002). Phase two also contained the greatest abundance of *Carpelimus* species, which are adapted for tunnelling in sand (Lott, 2009). Overall abundance of stenotopic staphylinids was greatest one year after scraping, partly because some species built up larger populations. 57.8% of stenotopic specimens in phase one were *Bledius gallicus*, compared with 5.3% in phase two. Adult *Bledius* also dig burrows in

fine sediments, where the larvae feed on algae (Staniec, 1999). Among other stenotopic species, *Planeustomus palpalis* was only recorded in phase one, and has previously been found in waterlogged turf (Steel, 1949).

Staphylinid assemblages at reference and control plots partly overlapped, with limited occurrence of stenotopic species. These areas were not winter-wet, and most of the stenotopic staphylinids in the present study have been associated with hydric conditions in previous research (Lott, 1998). *Xantholinus elegans* occurred at control, reference, and phase one plots. This species is often found in the early stages of ecological succession, but not limited to these (Lott and Anderson, 2011). Unlike the carabid assemblage, abundance of generalist staphylinids was significantly greater at control plots than all other treatments. Trees and canopied scrub had a similar effect in Spanish wood-pastures, supporting greater staphylinid abundance compared with grassland or scattered shrubs (García-Tejero and Taboada, 2016). Many species are associated with leaf litter, which has been identified as a critical resource in the taxonomic and ecological diversification of staphylinids (McKenna *et al.*, 2015). Staphylinids also occupy a wider range of environmental niches than carabids, including three species of *Gyrophaena* in the control plots that exploit saproxylic fungi (White, 1977).

2.4.4 Response of spiders

Abundance of stenotopic spiders was significantly greater in restored areas, compared with unmanaged plots. Succession to secondary woodland had a similar effect at coastal dunes, with a reduction of specialist species associated with sand dynamics (Bonte *et al.*, 2002). Although scraping benefited stenotopic spiders, it is likely that some of these depend on nearby patches of litter-rich vegetation for juvenile development, or as a retreat for adults (Bonte *et al.*, 2000). Abundance of generalist spiders was also significantly greater at control plots than all other treatments. This is consistent with reduced abundance of generalist spiders a year after turf stripping (Pedley *et al.*, 2013). Severe mechanical disturbance removes lower vegetation layers, which were a key determinant of epigeal spider diversity in saltmarshes (Pétillon *et al.*, 2008) and plantation forests (Oxbrough *et al.*, 2005). Litter at the ground surface also affected spiders through prey abundance, microclimate, and web attachment points (Uetz, 1979). The most common woodland-associated species in the

present study were *Diplocephalus picinus* and *Microneta viaria*, linyphiids which spin sheet webs among fallen leaves (Roberts, 1987).

Abundance and species richness of specialist spiders were not significantly greater after scraping in sandy grassland beside forestry trackways (Pedley *et al.*, 2013). Aerial dispersal of spiders is a passive mechanism ('ballooning' on strands of silk) (Bell *et al.*, 2005), so landscape configuration probably affects colonising spiders to a greater extent than carabids or staphylinids. This was demonstrated by the importance of nearby non-crop habitats as a source of immigrant spiders to arable fields (Schmidt *et al.*, 2008). It has also been suggested that in fragmented habitats, ballooning propensity declines in specialist spiders (Bonte *et al.*, 2003b). There were no stenotopic spider indicators for the individual restoration areas, although some species occurred in phases one and two. Cursorial lycosids were particularly abundant, including *Arctosa leopardus* (mean abundance 12.7 per plot in phase one, and 5.5 in phase two) and *Pardosa agrestis* (mean abundance 37.3 per plot in phase one, and 23.6 in phase two). *Arctosa leopardus* was an indicator species for newly restored wet heathland (Cristofoli *et al.*, 2010). *Pardosa agrestis* was characteristic of heath meadows in Finland (Koponen, 2000), although it was classified as an agrobiont in Hungary (Samu and Szinetár, 2002).

There was limited convergence between restoration and reference assemblages in the present study, although this was caused by differences in abundance rather than species composition. Only one of twelve stenotopic spider species in the restoration areas was not recorded at reference or control plots, so it is likely that local populations were important as a source of colonists. Stenotopic indicator species in restored areas preferred moister conditions, with lower values among preferences calculated for central European spiders (*Arctosa leopardus* 0.28; *Pardosa agrestis* 0.30) (Entling *et al.*, 2007). At reference plots, species were more associated with xeric conditions, based on the same index (*Drassyllus pusillus* 0.50; *Trachyzelotes pedestris* 0.94) (Entling *et al.*, 2007). No score was available for *Zodarion italicum*, although descriptions of ecological requirements for this species also suggest an association with dry, open habitats containing some bare ground (Harvey and Murphy, 1985). The structuring effect of moisture on spider communities is consistent in research from other sandy habitats, including

grassland (Buchholz, 2010), heathland (Schirmel and Buchholz, 2011) and coastal dunes (Bonte *et al.*, 2002).

2.4.5 Habitat impacts

Scraping and soil inversion are established conservation management techniques in habitats including chalk grassland (Edwards *et al.*, 2007), wet heaths (Godefroid *et al.*, 2016), sand dunes (Jones *et al.*, 2010) and dune slacks (Grootjans *et al.*, 2013). The persistence of disturbance effects varied in other research. Percentage cover of bare ground was 66% in dry sandy grassland one year after turf stripping (Pedley *et al.*, 2013), 26% one year after reprofiling alluvial dunes (Exeler *et al.*, 2009), and 46% two years after excavator disturbance (< 30 cm) in calcareous sandy grassland (Ödman *et al.*, 2011). In the present study, revegetation was more extensive and percentage cover of bare ground decreased to 11% (± S.E. 3.2) after one year. Possible explanations for these differences include varied disturbance techniques, and substrate chemistry (Dolman and Sutherland, 1994). Phosphorous limitation on plant growth may be reduced in soil under scrub, compared with semi-natural grassland (Gough and Marrs, 1990). Seedling recruitment could also have been enhanced in the restored areas, if moisture availability was higher after winter flooding (Maun, 1994).

The nature of colonising vegetation may be an additional cause of the rapid reduction in bare ground. Live roots remained in the soil after scraping, and common reed *Phragmites australis* was the dominant plant species in 5 of 12 quadrats, one year after disturbance. Extensive regeneration from *Phragmites australis* rhizomes was also observed after dune restoration in the Netherlands (Arens and Geelen, 2006). Creeping bent *Agrostis stolonifera* was the dominant plant species in six quadrats and probably colonised from seed rain (Hutchings and Booth, 1996). This suggests that scraping did not greatly reduce availability of nitrogen (Lammerts and Grootjans, 1997). The age of scrub may have a critical effect on seed bank composition, with limited resemblance to calcareous grassland under mature scrub established for more than 40 years (Bossuyt *et al.*, 2006). Availability of bare ground is of primary importance for some invertebrates, particularly as a nesting resource (Cornelisse *et al.*, 2013). Many other species respond to complex ecological interactions between substrate, vegetation, hydrology, and microclimate (Schirmel and Buchholz, 2011). As a

result, the area of bare ground could not be used to predict an appropriate frequency for disturbance in the present study.

The potential for plant invasions to alter long-term patterns of succession should be considered when planning any disturbance regime (D'Antonio and Meyerson, 2002). Among woody plants one year after scraping, *Betula pendula* was present in 9 of 12 quadrats. There was also *Betula* encroachment in reversion of arable land to sandy grassland, even with year-round grazing (Gilhaus *et al.*, 2015). This is consistent with tree invasions reported at inactive sand quarries (Řehounková and Prach, 2008) and presents a considerable management legacy. It is recommended that potential sources of wind-dispersed scrub seed are removed from adjacent areas, before scraping to bare substrate. The alien goat's-rue *Galega officinalis* was another colonist in the restored areas one year after scraping. This is a frequent invasive plant in the United States and Europe (Lastrucci *et al.*, 2010). Although only present in two quadrats, the monitoring plots did not adequately capture single plants scattered through the restored areas. A five-fold increase in frequency of *G. officinalis* was subsequently estimated in 2016. This plant has some value for bumblebees (Connop *et al.*, 2010) but can limit the availability of other forage, particularly in late summer (Robins, 2013).

2.4.6 Limitations

The present study was not based on a designed experiment, and subplots within treatments were pseudoreplicates (Hurlbert, 1984). Although no formal statistical distinction can be made from the effects of location, from interpretation of commuity ecology it is likely that major differences were the result of treatments. Positive effects of disturbance interventions for stenotopic invertebrates were also broadly similar to more typically investigated xeric grasslands and heathland (Pedley *et al.*, 2013; Olsson *et al.*, 2014). This suggests that the results are likely to have some validity outside the frame of Canvey Wick. Although the current research included three species-rich invertebrate groups, other taxa may require different components in a habitat mosaic. At post-industrial sites in Germany, northern lapwing *Vanellus vanellus* (Charadriidae) nested in recently disturbed areas, and chicks avoided aerial predators by retreating into dense willow scrub (Kamp *et al.*, 2015). To examine the potential significance of scrub for invertebrates, a preliminary investigation

of calypterate Diptera was completed at Canvey Wick (Appendix 9.2, page 297). The results suggest that further research is required, to ascertain effective configurations of scrub elements in a habitat mosaic.

2.4.7 Implications for restoration management

Most unmanaged early successional sites will undergo ecological succession, usually towards secondary woodland (Prach and Pyšek, 2001). Management techniques to reverse this have been examined in calcareous grassland, though without assessment of conservation value in the resulting carabid and spider communities (Rushton, 1988; Rushton *et al.*, 1990). The results of the present research demonstrated that despite the limited time since scraping, no individual stage of succession had a wholly representative complement of stenotopic species. Management prescriptions at Canvey Wick should include continuous availability of newly scraped areas, as part of the overall mosaic of successional stages available at the site (Kirby, 1992). Previous ground disturbance studies have not highlighted short-term species turnover for invertebrates. The dung beetle *Rhyssemus germanus* was also recorded once in the phase two restoration samples, which provides anecdotal evidence that other groups may benefit from the earliest stages of succession. This species was presumed extinct in the UK but has occurred at three other sites in south-east England since 2010 (Lane and Mann, 2016).

There was limited similarity between restoration and reference open mosaic habitat communities. Lack of convergence between restoration and target communities is not unusual in invertebrate research (Woodcock *et al.*, 2008). This does not represent failure for the management technique, as the developing communities have separate conservation value to existing areas of mosaic habitat. It also suggests that future restoration work should focus on manipulating the size and configuration of scrub blocks, although this is more difficult practically than grassland disturbance (Mortimer *et al.*, 2000). Contrasting habitat requirements of early successional invertebrates are consistent with previous studies (Cizek *et al.*, 2012) but in practice it is challenging to accommodate these within a heterogeneous disturbance regime (Warren *et al.*, 2007). Particular concerns are to establish a suitable frequency and scale for ground disturbance, based on data including medium and long-term effects. With the limited time series available, it was only possible to

speculate whether many species were breeding, might fail to establish, or occurred after accidental immigration.

Chapter 3 – Canvey Wick restoration: trait-based analysis

3.1 Introduction

The study of taxonomic composition has been central to the development of community ecology (Lawton, 1999). A limitation is that ecological information is lacking for many invertebrates, particularly where taxonomic knowledge is restricted (Langlands *et al.*, 2011). Community assembly also differs between ecological systems, which makes generalisation difficult. It is possible to extend analysis from the management community defined in Chapter 2, to include a broad range of species functional traits. These offer an alternative analytical framework to explain and predict community composition. Habitat templet theory describes the environment as a setting for the evolution of morphological traits, and as a filter for ecological sorting of species (Southwood, 1977). In landscapes extensively modified by human activity, it is likely that filtering is important, in addition to interactions within the community (Ribera *et al.*, 2001). Broad scale functional indicators offer a mechanistic understanding of the impact of environmental changes on biodiversity (Fournier *et al.*, 2015). Shared traits could permit integrated assessments in different bioregions, independent of taxonomic composition (McGill *et al.*, 2006).

Traits are morphological, phenological, physiological and ecological attributes that potentially influence species performance or fitness (Fountain-Jones *et al.*, 2015). These include direct measures of performance such as fecundity, or functions that indirectly affect reproduction, growth, or persistence in the environment (Violle *et al.*, 2007). It is possible to describe multiple traits in a single continuous diversity index, with different measures to assess trait values, spread, or distribution (Spake *et al.*, 2016). These have been successfully used to represent ecosystem services, and are not necessarily correlated with taxonomic measures of species richness or diversity (Gagic *et al.*, 2015). Forest spider assemblages exhibited increased functional diversity despite lower species richness, suggesting complementary traits for prey capture and consumption (Schuldt *et al.*, 2014). Assemblages with low functional diversity may be more likely to exhibit niche overlap and interference competition, although duplication of traits could benefit long-term ecosystem stability (Woodcock *et al.*, 2010).

Trait-based approaches were initially developed in vegetation ecology (Lavorel and Garnier, 2002) and have recently been used to characterise community patterns and impacts of human activity for invertebrates. Anthropogenic environmental change is frequently linked to trophic shifts, such as landscape simplification and a decline of predatory carabids (Purtauf *et al.*, 2005; Gobbi and Fontaneto, 2008). Sensitivity of carabids to fragmentation of herbaceous and shrub-dominated habitats was partly explained by phenological traits, including spring adult activity and summer larval period (Barbaro and van Halder, 2009). Ecological traits are complex consequences of individual adaptations (Fountain-Jones *et al.*, 2015). Moisture and shade tolerance combine morphological and physiological traits, and were important drivers for beta-diversity of spiders, especially in open habitats (Entling *et al.*, 2007). In addition to response traits, ecosystem effects have been demonstrated for some invertebrates (Prather *et al.*, 2013; Gagic *et al.*, 2015). Graminivorous carabids affected plant recruitment as dominant consumers of *Taraxacum* seeds (Honek *et al.*, 2009), and carabids and lycosids were important epigeal arthropod predators in agricultural systems (Lang *et al.*, 1999).

The selection of traits for investigation is critical, because these form the basis of subsequent classifications and calculations of ecological similarity (Weithoff, 2003). This should follow from presupposed links to functional or ecological significance in the community assembly process (McGill *et al.*, 2006). This prioritisation makes it possible to hypothesise about patterns, underpinned by rules and mechanisms (Lawton, 1999). The process is not straightforward, in part because evolutionary relationships create autocorrelation in trait data (Fountain-Jones *et al.*, 2015). Phylogeny and morphology shared power to predict beetle diet and microhabitat use, although there was substantial variation in the expression of functional traits (Barton *et al.*, 2011). It may also be difficult to disentangle potential links between ecological and functional attributes, as with the association of larger cursorial spiders with dispersal, and running hunting strategy (Pedley and Dolman, 2014). For this reason, it is important to select traits based on the anticipated effects of environmental pressures being studied.

Disturbance regimes have been linked to trait shifts in some invertebrate communities. Larger carabids declined in regularly disturbed open and

grassland habitats, particularly species with low fecundity and longer larval periods (Kotze and O'Hara, 2003). Increasing intensity of land management was associated with smaller body size and greater frequency of macropterous carabids (Ribera *et al.*, 2001). Pterygomorphism is directly linked to dispersal capability, allowing species to exploit patchy resources at the local and landscape scales (den Boer, 1990). Disturbance may affect other functional traits by altering succession, vegetation structure, microclimate and resource availability (Schirmel *et al.*, 2015). Bare ground enhances invertebrate thermoregulation (Honek, 1988) and body size for European spiders increased in warm, arid environments (Entling *et al.*, 2010). There was a shift in dominance from predatory to phytophagous carabids along an experimental ground disturbance gradient (Pedley and Dolman, 2014). In a fire disturbance regime, burrow ambush and burrowing were more frequent prey capture strategies for spiders at recently burnt sites (Langlands *et al.*, 2011).

The extent to which it is possible to develop a predictive understanding of disturbance from trait-environment relationships is unclear. Complementary and contrasting trait responses have been reported between and within taxonomic groups. Increased flood intensity was positively associated with propensity for ballooning in spiders (Lambeets *et al.*, 2008). The same effect was not apparent at different stages of succession following ground disturbance, although increasing body size of cursorial species was interpreted as a trait favouring local dispersal (Pedley and Dolman, 2014). This may be an effect of different disturbance regimes, or studies of experimental and long-established landscape gradients (Moretti *et al.*, 2006). Divergence has also been observed within taxonomic groups, with smaller lycosid spiders associated with flooding disturbance, compared with larger linyphiids (Lambeets *et al.*, 2008). Similarly, at the level of functional diversity, studies have suggested both positive (Pakeman, 2014) and negative relationships with disturbance (Gerisch *et al.*, 2012; Schirmel *et al.*, 2012).

The predictive capacity of a common set of traits must be established over a wide range of conditions (de Bello *et al.*, 2005). The variation reported in invertebrate community assembly following disturbance suggests that further research is required. Brownfields are a suitable environment for this, possessing a specialised early successional fauna (Shaw, 2011). Lack of

management can lead to shifts in vegetation structure that affect habitat suitability for species adapted to pioneer conditions and sporadic disturbance (Lambeets *et al.*, 2008). Congruence between functional indices and traits representing species of regional conservation value is also relevant in this context. Extending a trait-based framework to staphylinids would be valuable, as these are less well studied than carabids or spiders. A broader range of taxa could improve mechanistic understanding of underlying processes (Aubin *et al.*, 2013).

This study evaluates invertebrate functional responses to resetting ecological succession at a sandy brownfield colonised by scrub and trees. The community assembly process was followed in different areas, cleared in the previous two winters. It was expected that functional traits would be influenced by time since clearance, and only be partly congruent between different invertebrate groups. To explore this, the following questions were asked. (1) What are the effects on community trait composition of carabids, staphylinids and spiders? (2) Are there common patterns of response to disturbance across taxonomic groups? (3) How does functional diversity change with time after disturbance? (4) How does removal of vegetation and soil organic matter compare with previously reported trait responses to disturbance?

3.2 Methods

3.2.1 Experimental design

Analyses for this study were based on the samples collected during monitoring of restoration management at Canvey Wick in 2015. The experimental design and invertebrate sampling were the same as reported in Chapter 2.

3.2.2 Invertebrate classification – traits

Traits were collated for 246 taxa, comprising 80 carabid, 99 staphylinid, and 67 spider species. Trait selection was based on potential functional links to disturbance, identified in previous research (Ribera *et al.*, 2001; Lambeets *et al.*, 2008; Langlands *et al.*, 2011; Schirmel *et al.*, 2012; Pedley and Dolman, 2014). For each invertebrate group, traits related to morphology, phenology and ecological performance (Fountain-Jones *et al.*, 2015) (Table 3.1), although it is recognised that some traits could be linked to multiple ecological functions. It is

Table 3.1 Traits collated for Carabidae (Coleoptera) [a], Staphylinidae (Coleoptera) [b], and Araneae [c] recorded in management areas at Canvey Wick in 2015. Species-level trait coding is detailed in Appendix 9.3 (page 303).

Trait type	Trait	Trait state
Morphological	Size [a b c]	Body length
	Dispersal [a c]	Apterous or brachypterous [a] Wing-dimorphic [a] Macropterous [a] Ballooning [c]
Phenological	Breeding season [a]	Spring Autumn
Ecological performance	Adult trophic guild [a]	Predatory Omnivorous Phytophagous
	Adult resource preference [b]	Litter Dung or carrion Mycetophilous Myrmecophilous Phytophagous
	Adult hunting strategy [c]	Ambush hunter Running hunter Web hunter
	Environmental tolerance [c]	Moisture Shade
	Conservation status [a b c]	UK designation

possible that disturbance might have selected for smaller individuals in some species, although individual trait variation for arthropods may be less than variation across all sampled species (Simons *et al.*, 2016). Conservation status is nationally specific but was included for comparison with functional diversity indices.

For staphylinids, body size was the median length in millimetres from the range given in literature (Lott, 2009; Lott and Anderson, 2011; Freude *et al.*, 1974). Limited information was available about ecological traits, particularly the distinction between adult and larval diet (Good and Giller, 1991). In the present study, staphylinids were classified by adult resource preference (García-Tejero and Taboada, 2016) as litter-dwelling, frequently associated with dung and carrion, phytophagous, mycetophilous, or myrmecophilous. *Pella limbata* was the only probable specialist ant predator (Stoeffler *et al.*, 2011). *Drusilla canaliculata* is defensively adapted for myrmecophily (Brand *et al.*, 1973) but is usually treated as a generalist predator (Balog *et al.*, 2013). Staphylinid conservation status was taken from the most recent published review (Hyman and Parsons, 1994).

For carabids, body size was the median length in millimetres from the range given in Luff (2007). Breeding season was classified as spring or autumn (Luff, 2007). Where this information was not available, inference was made from the overwintering stage as larva (autumn), or adult (spring). Wing development was divided into three categories: brachypterous, dimorphic, and macropterous (Luff, 2007). Apterous species were included in the brachypterous category as both morphs are unable to fly. Some macropterous individuals may lack developed flight muscles, although these are probably reabsorbed by histolysis prior to reproduction (Roff, 1986). Carabid trophic strategies were classified as predatory, omnivorous, or graminivorous. This is potentially problematic, due to the distinction between obligate and frequently graminivorous species (Talarico *et al.*, 2016). For comparison with most recent trait studies, the broader concept of frequently graminivorous species within the Harpalini was adopted (Vanbergen *et al.*, 2010). Carabid conservation status was taken from the most recent published review (Telfer, 2016).

For spiders, body size was the median length in millimetres from the range given in literature for females (Roberts, 1985; Roberts, 1987). Data for moisture

and shade preference were also available (Entling *et al.*, 2007). Species were scored as ballooners if there was a species-specific record of aerial dispersal using silk (Bell *et al.*, 2005; Blandenier, 2009). It is recognised that this does not incorporate the differing frequency of ballooning among species, or effectiveness for dispersal which may be limited to a few metres in some genera (Morse, 1993). Prey capture strategy was classified as running hunter, ambush hunter, or web hunter (Uetz *et al.*, 1999). Due to the reliance on pitfall trapping in this study, all classes of web building spiders were not effectively sampled, so this category was not subdivided. A lack of specimens in active hunting categories meant that it was also not possible to distinguish between ground or foliage running hunters, or stalking and ambush hunters. Spider conservation status was taken from the most recent available review (Merrett, 1990).

3.2.3 Statistical analysis

Principal Coordinates Analysis was completed with the Canoco 5.0 software package (ter Braak and Šmilauer, 2014). All other analyses were completed in the statistical software R (R Core Team, 2017). Precise *p*-values are reported, to allow interpretation of *p* as a measure of evidence, and to facilitate meta-analysis (Cumming and Calin-Jageman, 2017).

3.2.3.1 Principal Coordinates Analysis

Correlation among traits was examined for each taxonomic group by Principal Coordinates Analysis (PCoA). This approach was necessary as the trait data included quantitative and qualitative measures. The dissimilarity matrix was calculated based on the Gower coefficient, with species as samples and traits as variables (Gower, 1971; Podani, 1999). The axes in PCoA maximise the linear correlation between distances in the distance matrix, and the distances in a two-dimensional plane. To visualise associations, traits were plotted as vectors, based on loadings relative to the first two PCoA axes (Langlands *et al.*, 2011). The lengths of the vectors are proportional to their correlation coefficient, and the cosine of the angle between trait vectors approximates the correlation of those traits (Liebergesell *et al.*, 2016). Where traits are strongly correlated, it must be recognised that it is not possible to establish a direct functional link to species occurrence. To visualise the influence of phylogeny on the distribution

of traits, species were grouped by the six most frequent families (spiders), or tribes (carabids and staphylinids).

3.2.3.2 Functional indices

Trait diversity is a measure of the differences between traits within a species assemblage, and as a latent variable can only be measured by one or more indices. Functional evenness, functional dispersion and functional divergence were calculated, as these represent distinct components of trait diversity (Weiher, 2010). Functional dispersion describes variation in species trait composition, as the mean distance to the centroid of the assemblage in trait space, weighted by relative abundances (Laliberté and Legendre, 2010). Functional evenness measures the extent to which functional niches are evenly exploited within a community, from the consistency in branch lengths of the minimum spanning tree connecting all species in trait space (Mason *et al.*, 2005). Functional divergence examines the spread of functional traits (Villéger *et al.*, 2008), as the tendency of the most abundant species to be at the periphery of the assemblage in trait space. Functional distances between species were calculated using Gower's distance metric (Gower, 1971; Podani, 1999). This allows for a mixture of continuous, ordinal, and categorical variables, and can accommodate missing trait values.

Functional metrics were calculated separately for carabids and spiders at each monitoring plot. Indices were weighted by abundance, which has been demonstrated to improve explanation of ecological functions compared with presence-absence data (Gagic *et al.*, 2015). Although most multi-trait functional diversity indices were weakly affected by trait choice (Gagic *et al.*, 2015), staphylinid beetles were not included because of the limited number of traits compiled for this family (Rusch *et al.*, 2015). It was not possible to compare index values for treatments with a generalized linear model, as model residuals were strongly non-normal. Comparisons were completed with the Kruskal–Wallis (H) test (Kruskal and Wallis, 1952), followed by Dunn's test to report results among multiple pairwise comparisons (Dunn, 1964). To adjust p-values, the family-wise error rate was controlled using the Holm-Šidák adjustment (Holm, 1979) in the dunn.test package (Dinno, 2016). This is a progressive step-up procedure assuming dependence between tests, which are completed in order of increasing probability α/n-1. This was preferred to the classical

Bonferroni correction, with tests compared to α/n, which gives a smaller or equal probability of rejecting false hypotheses for the same data (Holm, 1979).

3.2.3.3 Fourth corner analysis

The relationships between species functional traits and environmental parameters were examined separately for each taxonomic group by 'fourth corner' permutation analysis (Legendre *et al.*, 1997). This relates matrices for environmental variables (R) against species traits (Q) through species occurrence (L). Emergent group analysis has been presented as a complement to the fourth-corner approach (Langlands *et al.*, 2011). This may suggest functional types (Keith *et al.*, 2007), although there were too few staphylinid traits to attempt this. Matrix R contained the four habitat treatments for wooded control plots, early successional reference plots, newly scraped plots, and plots scraped one year previously. Matrix Q contained the species traits described in Table 3.1. Matrix L contained the pooled abundance for each species per monitoring plot, Hellinger transformed to better reflect the relative contributions of abundant and rare species to the analysis (Legendre and Gallagher, 2001). A test statistic was calculated separately for each pair of treatments and species traits. Treatments were qualitative, and where traits were quantitative a correlation ratio (Pseudo-*F*) was calculated. Where traits were qualitative a Pearson statistic (χ^2) was calculated, divided by the sum of abundances in the species table.

A permutation test was applied to determine correlation. This was based on 'Model 1' which presupposes that individuals of a species are distributed where they encounter appropriate site conditions (Legendre *et al.*, 1997). Links between traits of species (fixed) and characteristics of sampling sites (fixed) are mediated by the abundance data (random) (Aubin *et al.*, 2013). In these circumstances, Model 1 can be used to test a null hypothesis that environmental variables are not linked to species traits, at the level of individual species (Dray and Legendre, 2008). This produces correct rates of Type I error, although a combination of permutation models 2 and 4 is required when tables R or Q are random (ter Braak *et al.*, 2012). *P*-values were adjusted to account for false discovery rates in multiple testing of environmental parameters and species traits (Verhoeven *et al.*, 2005). No phylogenetic correction was applied,

as species-level phylogenetic trees were unavailable for all beetles and spiders (Pedley and Dolman, 2014).

3.3 Results

3.3.1 Principal Coordinates Analysis

3.3.3.1 Carabidae

The first two axes of the PCoA for carabids explained 56.1% of the variation between species, based on the five traits (Figure 3.1, Table 3.2). PCoA axis one represented 30.3% of trait variation, particularly wing morphology and breeding season. PCoA axis two represented 25.8% of trait variation, particularly trophic group. The highest scores for variable loadings on axis one were for autumn breeding, and brachypterous wings. The lowest scores were for spring breeding and macropterous wings. The highest score for variable loading on axis two was for phytophagous trophic group. The lowest score was for predatory trophic group. Several traits were correlated, particularly larger body size with autumn breeding. The biplot also showed the influence of phylogeny on traits, including the tendency for Harpalini to be phytophagous.

3.3.3.2 Staphylinidae

The first two axes of the PCoA for staphylinids explained 73.6% of the variation between species, based on the three traits (Figure 3.2, Table 3.2). PCoA axis one represented 58.1% of trait variation, particularly resource preference. PCoA axis two represented 15.6% of trait variation, particularly phytophagous resource preference. The highest score for variable loading on axis one was for dung or carrion resource preference. The lowest score was for litter resource preference. The highest score for variable loading on axis two was for phytophagous resource preference. Overall there was limited correlation between traits. The trait classification lacks sufficient depth for the biplot to clearly show the influence of phylogeny on traits, although the development of phytophagy in the Thinobiini is evident.

3.3.3.3 Araneae

The first two axes of the PCoA for spiders explained 65.7% of the variation between species, based on the six traits (Figure 3.3, Table 3.2). PCoA axis one

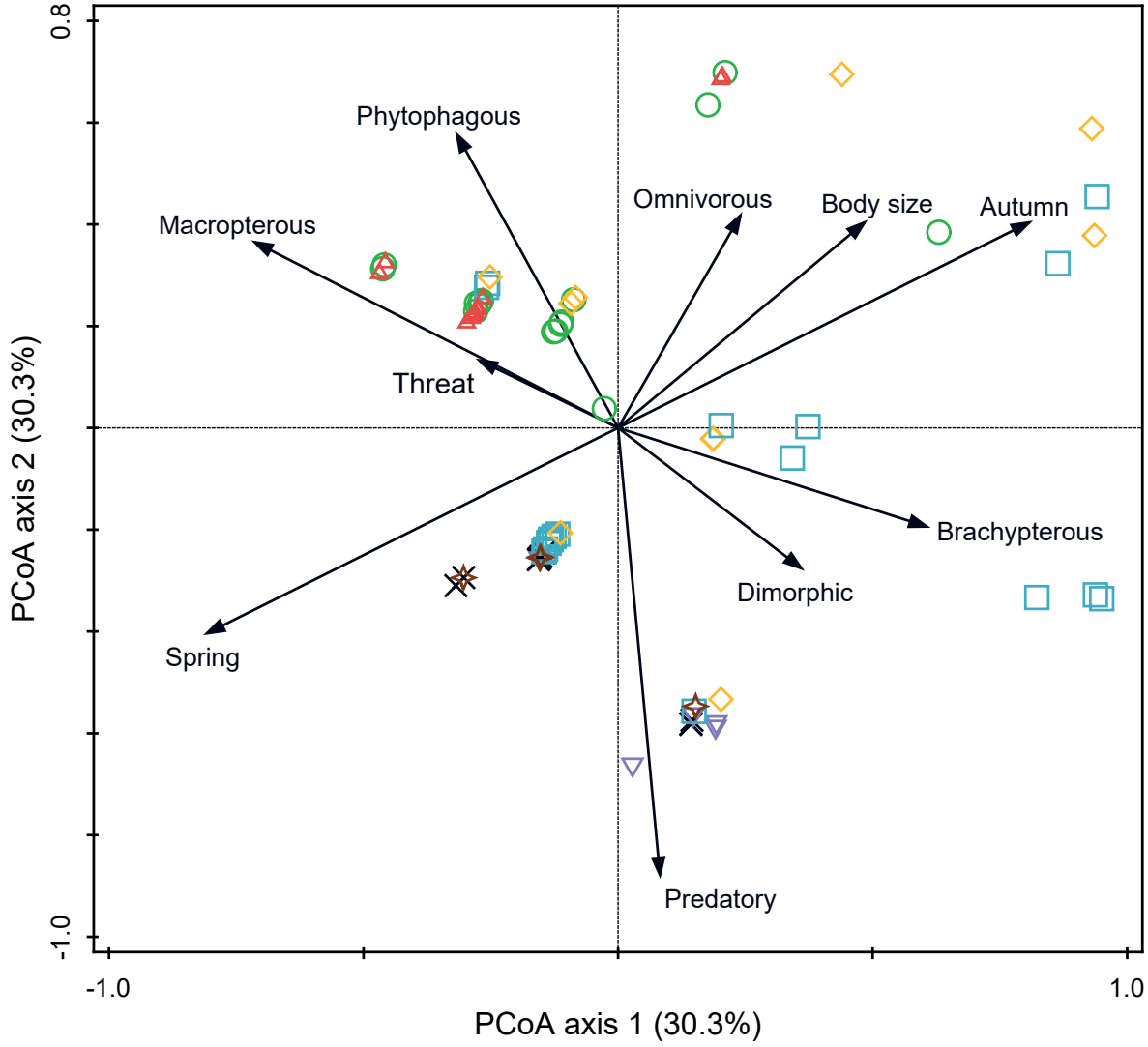

Figure 3.1 Principal Coordinates Analysis (PCoA) of the similarity between 80 species of Carabidae, based on 10 traits. Wing morphology is given as macropterous, dimorphic, or brachypterous. Trophic group is given as predatory, omnivorous, or phytophagous. Breeding period is given as spring or autumn. Body size is the median from the range given in Luff (2007). Conservation status is based on the most recent published review (Telfer, 2016). Vectors represent the Spearman correlation of traits, with the length and direction of each arrow indicating the relationship with PCoA axes. Samples were collected from Canvey Wick, Essex, in 2015 from areas of closed canopy scrub (n=6), scrub cleared and substrate scraped to mineral soil in May 2014 (phase one, n=3), scrub cleared and substrate scraped to mineral soil in May 2015 (n=6), and open mosaic habitat reference plots (n=5). Samples were pooled in each year from six pitfall traps per monitoring plot, open for nine days in June, July, and August. Species within the six most numerically abundant carabid tribes are displayed with symbols: Bembidiini (✕), Harpalini (○), Lebiini (▽), Pterostichini (◇), Scaritini (✦), and Zabrini (△). The remaining species are pooled into one symbol (□).

Table 3.2 Trait loadings for Carabidae (Coleoptera), Staphylinidae (Coleoptera), and Araneae, based on axes one and two scores derived from Principal Coordinates Analysis (PCoA). Scores exceeding 0.5/-0.5 are shown in bold.

Invertebrate group	Trait	PCoA Axis 1	PCoA Axis 2
Coleoptera: Carabidae	Body size	0.4842	0.4037
	Spring breeding	**-0.8077**	-0.4037
	Autumn breeding	**0.8077**	0.4037
	Predatory	0.0821	**-0.8783**
	Omnivorous	0.2402	0.4174
	Phytophagous	-0.3165	**0.5766**
	Brachypterous	**0.6062**	-0.1938
	Dimorphic	0.3600	-0.2759
	Macropterous	**-0.7136**	0.3650
	Threat	-0.2727	0.1328
Coleoptera: Staphylinidae	Body size	-0.3209	-0.1639
	Dung/carrion	**0.6555**	-0.4923
	Litter	**-0.9142**	0.0306
	Mycetophilous	0.4509	0.0941
	Myrmecophilous	0.1624	-0.0138
	Phytophagous	0.1693	**0.8403**
	Threat	-0.4806	-0.2981
Araneae	Body size	**0.6979**	0.0228
	Moisture	0.4754	-0.1810
	Shade	-0.0353	-0.2269
	Ballooning	-0.3284	**0.9269**
	Web builder	**-0.9235**	-0.1135
	Running hunter	**0.8934**	-0.1925
	Ambush hunter	0.1313	-0.0965
	Threat	0.4606	0.2241

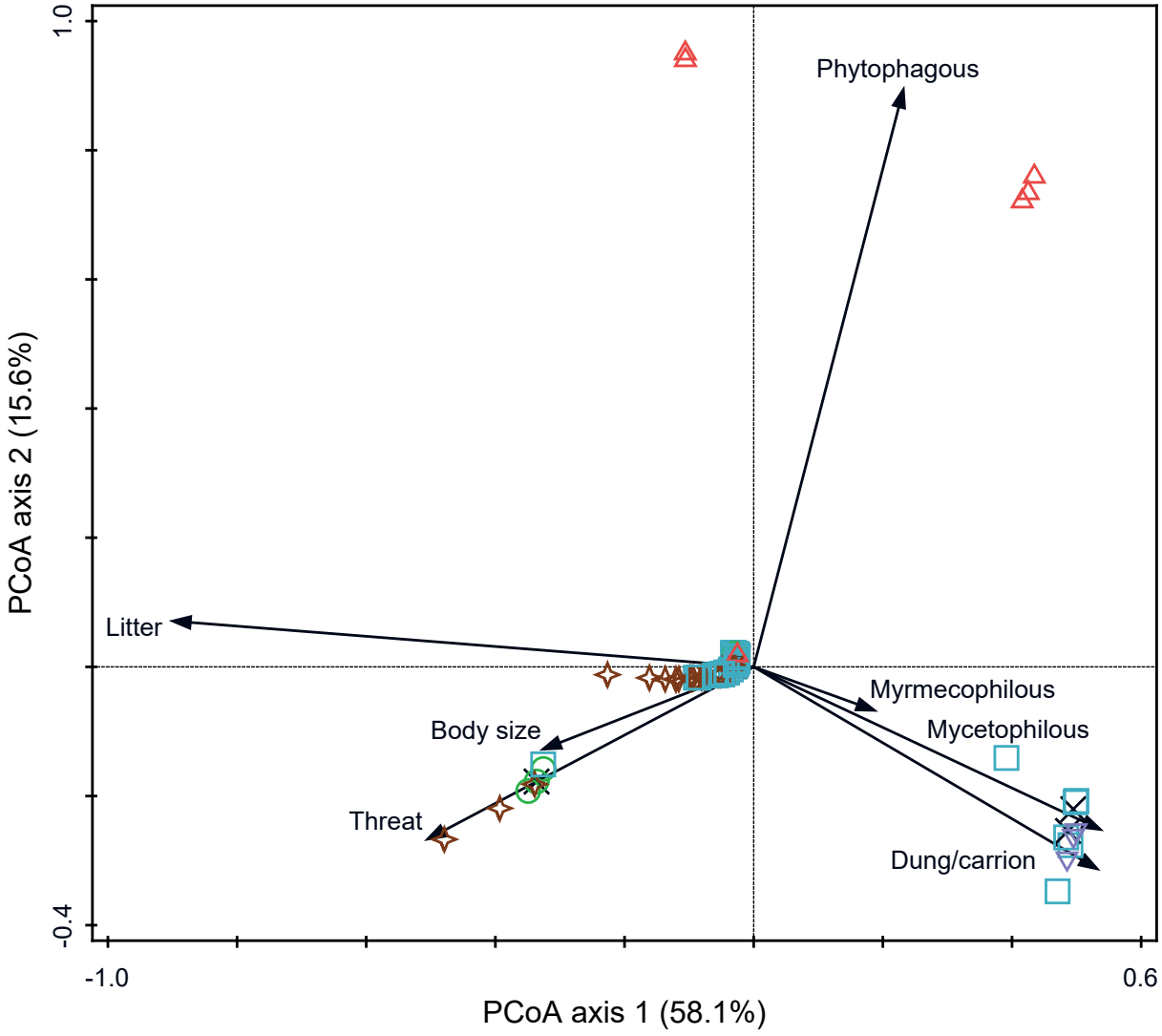

Figure 3.2 Principal Coordinates Analysis (PCoA) of the similarity between 99 species of Staphylinidae, based on three traits. Body size is the median from the range given in Lott (2009), Lott & Anderson (2011), and Freude *et al*. (1974). Adult resource preference is given as litter, myrmecophilous, phytophagous, mycetophilous, or dung and carrion. Conservation status is based on the most recent published review (Hyman, 1994). Vectors represent the Spearman correlation of traits, with the length and direction of each arrow indicating the relationship with PCoA axes. Samples were collected from Canvey Wick, Essex, in 2015 from areas of closed canopy scrub (n=6), scrub cleared and substrate scraped to mineral soil in May 2014 (phase one, n=3), scrub cleared and substrate scraped to mineral soil in May 2015 (n=6), and open mosaic habitat reference plots (n=5). Samples were pooled in each year from six pitfall traps per monitoring plot, open for nine days in June, July, and August. Species within the six most numerically abundant staphylinid tribes are displayed with unique symbols: Athetini (✕), Oxypodini (○), Oxytelini (▽), Paederini (◇), Staphylinini (✦) and Thinobiini (△). The remaining species are pooled into one symbol (□).

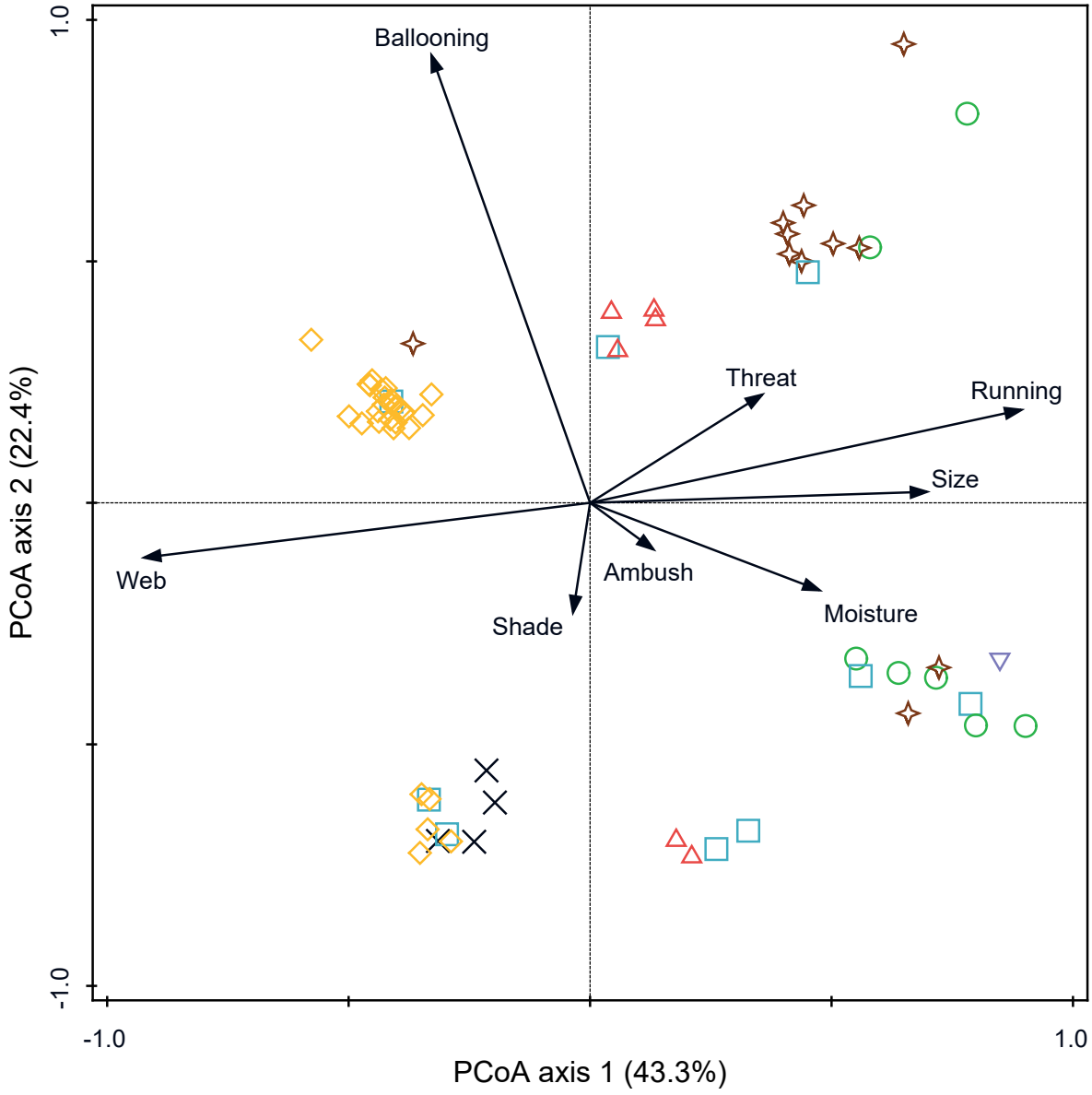

Figure 3.3 Principal Coordinates Analysis (PCoA) of the similarity between 67 species of spider, based on six traits. Hunting strategy is given as running, ambush, or web, based on the guild structure in Uetz (1999). Dispersal is based on species-specific records of ballooning in Bell (2005) and Blandenier (2009). Moisture and shade perference is taken from Entling (1997). Body size is the median from the range given for females in Roberts (1985, 1987). Conservation status is based on the most recent available review (Merrett, 1990). Vectors represent the Spearman correlation of traits, with the length and direction of each arrow indicating the relationship with PCoA axes. Samples were collected from Canvey Wick, Essex, in 2015 from areas of closed canopy scrub (n=6), scrub cleared and substrate scraped to mineral soil in May 2014 (phase one, n=3), scrub cleared and substrate scraped to mineral soil in May 2015 (n=6), and open mosaic habitat reference plots (n=5). Samples were pooled in each year from six pitfall traps per monitoring plot, open for nine days in June, July, and August. Species within the six most numerically abundant spider families are displayed with unique symbols: Clubionidae (✕), Gnaphosidae (○), Liocranidae (▽), Linyphiidae (◇), Lycosidae (✧), and Thomisidae (△). The remaining species are pooled into one symbol (□).

represented 43.3% of trait variation, particularly body size. PCoA axis two represented 22.4% of trait variation, particularly environmental tolerance. The highest scores for variable loadings on axis one were for larger body size and running hunters. The lowest score was for web-building. For axis two the highest score was for ballooning. Several traits were correlated, including running hunters with larger body size. In general, species associated with ballooning had lower shade tolerance, and web-builders were smaller in body size. The biplot also showed the influence of phylogeny on traits, including the tendency for Linyphiidae to have smaller body size, and greater shade tolerance for Clubionidae.

3.3.2 Functional metrics

Functional dispersion in the carabid data differed significantly between treatments ($\chi^2_{[3, N=20]} = 12.54$, $P = 0.0057$) (Table 3.3). This was significantly higher in control plots than phase two ($P_{adj} = 0.0019$) or reference plots ($P_{adj} = 0.0423$). Functional evenness did not differ significantly between all treatments ($\chi^2_{[3, N=20]} = 7.78$, $P = 0.0509$), although this was significantly higher at control plots than phase one plots ($P_{adj} = 0.0357$). There was no significant difference in functional divergence between treatments ($\chi^2_{[3, N=20]} = 6.01$, $P = 0.111$).

Functional dispersion in the spider data differed significantly between treatments ($\chi^2_{[3, N=20]} = 13.85$, $P = 0.0031$) (Table 3.3). This was significantly higher in control plots than phase two plots ($P_{adj} = 0.001$). Functional evenness differed significantly between treatments ($\chi^2_{[3, N=20]} = 13.41$, $P = 0.0038$). This was significantly higher in reference plots than phase one ($P_{adj} = 0.0172$) or phase two plots ($P_{adj} = 0.004$). Functional divergence differed significantly between treatments ($\chi^2_{[3, N=20]} = 15.43$, $P = 0.0015$). This was significantly higher in control plots than phase two plots ($P_{adj} = 0.0003$).

3.3.3 Fourth corner analysis

3.3.3.1 Carabids

The correlation matrix of classes of carabid traits by classes of treatment detected 12 significant associations. Spring breeding was negatively associated with control plots ($\chi^2 = -2.88$, $P_{adj} = 0.0196$). Phytophagous carabids were positively associated with phase one ($\chi^2 = 3.61$, $P_{adj} = 0.0018$) and reference

Table 3.3 Median (minimum – maximum) values, and results of Kruskal–Wallis (H) test values, comparing functional dispersion, functional evenness, and functional divergence among management areas at Canvey Wick in 2015. Treatments were unmanaged controls (n=6), phase one cleared to mineral substrate in May 2014 (n=3), phase two cleared to mineral substrate in May 2015 (n=6), and open mosaic habitat reference plots (n=5). Samples were pooled from six pitfall traps within each monitoring plot, open for nine days in June, July, and August 2015. Post-hoc tests show Tukey pairwise comparisons after Holm-Šidák adjustment, treatments with shared superscript a-c (ranked highest to lowest) did not differ significantly ($P > 0.05$).

Invertebrate group	Variable	Control	Phase One	Phase Two	Reference	Kruskal-Wallis$_{[3, N=20]}$
Carabidae	Functional dispersion	0.29 (0.27-0.31) ^{a}	0.22 (0.21-0.25) ^{ab}	0.20 (0.07-0.23) ^{b}	0.23 (0.12-0.28) ^{b}	H = 12.54
	Functional evenness	0.64 (0.50-0.68) ^{a}	0.48 (0.46-0.54) ^{b}	0.54 (0.50-0.57) ^{ab}	0.51 (0.48-0.66) ^{ab}	H = 7.78
	Functional divergence	0.93 (0.87-0.95) ^{a}	0.90 (0.86-0.91) ^{a}	0.92 (0.88-0.96) ^{a}	0.83 (0.73-0.95) ^{a}	H = 6.01
Araneae	Functional dispersion	0.25 (0.23–0.26) ^{a}	0.16 (0.16–0.21) ^{ab}	0.15 (0.12–0.20) ^{b}	0.20 (0.16-0.21) ^{ab}	H = 13.85
	Functional evenness	0.71 (0.58–0.74) ^{ab}	0.59 (0.51–0.60) ^{b}	0.56 (0.51–0.61) ^{b}	0.74 (0.70-0.79) ^{a}	H = 13.41
	Functional divergence	0.88 (0.85–0.91) ^{a}	0.81 (0.74–0.86) ^{ab}	0.70 (0.65–0.77) ^{b}	0.83 (0.80-0.86) ^{ab}	H = 15.43

plots (χ^2 = 3.81, P_{adj} = 0.0044), and negatively associated with control plots (χ^2 = -2.80, P_{adj} = 0.0189). Predatory carabids were negatively associated with control plots (χ^2 = -3.16, P_{adj} = 0.0041). Macropterous species were negatively associated with control plots (χ^2 = -3.18, P_{adj} = 0.0158) and positively associated with phase one plots (χ^2 = 5.60, P_{adj} = 0.0044). Wing dimorphic species were negatively associated with control plots (χ^2 = -2.76, P_{adj} = 0.0137) and positively associated with phase two plots (χ^2 = 2.57, P_{adj} = 0.0222). Brachypterous species were positively associated with reference plots (χ^2 = 2.46, P_{adj} = 0.0477). Species without conservation status were negatively associated with control plots (χ^2 = -2.80, P_{adj} < 0.0211). Species with conservation status were positively associated with phase one plots (χ^2 = 3.90, P_{adj} = 0.0132) and negatively associated with control plots (χ^2 = -2.44, P_{adj} = 0.0222).

3.3.3.2 Staphylinids

The correlation matrix of classes of staphylinid traits by classes of treatment detected 10 significant associations. Smaller body size was associated with phase two plots (Pseudo F = -2.27, P_{adj} = 0.0115). Resource preference for litter was positively associated with control plots (χ^2 = 5.86, P_{adj} = 0.0016) and negatively associated with reference plots (χ^2 = -3.85, P_{adj} = 0.0026). Resource preference for dung or carrion was negatively associated with control plots (χ^2 = -3.03, P_{adj} = 0.0064), and positively associated with phase two plots (χ^2 = 5.68, P_{adj} = 0.0016). Mycetophilous (χ^2 = 3.48, P_{adj} = 0.0078) and myrmecophilous species (χ^2 = 2.98, P_{adj} = 0.0064) were positively associated with control plots. Phytophagous species were positively associated with phase one plots (χ^2 = 4.82, P_{adj} = 0.0016). Species with conservation status were positively associated with phase two plots (χ^2 = 3.47, P_{adj} = 0.0053). Species without conservation status were negatively associated with reference plots (χ^2 = -4.79, P_{adj} = 0.0016).

3.3.3.3 Spiders

The correlation matrix of classes of spider traits by classes of treatment detected 24 significant associations. Body size was positively associated with control plots (Pseudo F = 3.71, P_{adj} = 0.0061). Moisture tolerance was

negatively associated with control (Pseudo F = -3.17, P_{adj} = 0.0005), reference (Pseudo F = -3.17, P_{adj} = 0.0005) and phase two plots (Pseudo F = -3.17, P_{adj} = 0.0005). Shade tolerance was positively associated with control plots (Pseudo F = 1.91, P_{adj} = 0.0446), and negatively associated with phase two (Pseudo F = -4.76, P_{adj} = 0.0002) and reference plots (Pseudo F = -4.01, P_{adj} = 0.0011). Species without conservation status were negatively associated with control (χ^2 = -2.41, P_{adj} = 0.023) and reference plots (χ^2 = -3.31, P_{adj} = 0.0012), and positively associated with phase one (χ^2 = 3.70, P_{adj} = 0.0012) and phase two plots (χ^2 = 2.92, P_{adj} = 0.0015).

No record of dispersal by ballooning was positively associated with control plots (χ^2 = 4.05, P_{adj} = 0.0011) and negatively associated with phase two plots (χ^2 = -2.70, P_{adj} = 0.0132). Dispersal by ballooning was negatively associated with control (χ^2 = -3.27, P_{adj} = 0.0021) and reference plots (χ^2 = -3.71, P_{adj} = 0.0012) and positively associated with phase one (χ^2 = 4.31, P_{adj} = 0.0011) and phase two plots (χ^2 = 3.47, P_{adj} = 0.0012). Ambush hunting strategy was positively associated with phase one plots (χ^2 = 3.97, P_{adj} = 0.0011) and negatively associated with phase two plots (χ^2 = -2.70, P_{adj} = 0.0097). Running hunting strategy was positively associated with phase one plots (χ^2 = 2.96, P_{adj} = 0.0093). Web hunting strategy was negatively associated with control (χ^2 = -2.43, P_{adj} = 0.023) and reference plots (χ^2 = -3.46, P_{adj} = 0.0017) and positively associated with phase one (χ^2 = 2.89, P_{adj} = 0.0071) and phase two plots (χ^2 = 3.52, P_{adj} = 0.003).

3.4 Discussion

This study examined trait responses in carabid, staphylinid and spider assemblages, up to one year after removal of vegetation and soil organic matter from an area of closed canopy scrub. Functional metrics were lower in disturbed areas. Each taxonomic group showed trait shifts, with potential functional links to resource utilisation, dispersal, and conservation status. Carabid traits linked to disturbance were wing morphology, trophic group and conservation status. Staphylinid traits were body size, resource preference and conservation status. Spider traits were body size, ballooning propensity and prey capture strategies. It was possible to interpret different traits in each group as adaptations for dispersal. There were also contrasting effects, with smaller staphylinids and

spiders associated with disturbance, and no effect on body size for carabids. As with the analysis in Chapter 2, subplots within treatments were pseudoreplicates (Hurlbert, 1984). As a result, no formal statistical distinction can be drawn between the effects of treatments and location on species traits.

3.4.1 Carabid response traits

Phytophagous carabids were less frequent at control plots, and more frequent in the phase one restoration, and early successional reference plots. This was probably linked to resource availability in forb species richness (Table 2.1, Chapter 2), which has been demonstrated experimentally (Harvey *et al.*, 2008). The density of phytophagous carabids was also greater in rotational cropping and grassland habitats, compared with forests at the landscape scale (Vanbergen *et al.*, 2010). Frequency of phytophagous carabids was not enhanced in phase two, with forb species richness similar to phase one but perhaps lacking sufficient vegetation cover for predator avoidance (Kaspari and Joern, 1993). Abundance of predatory carabids was reduced at control plots. Predatory carabids were more frequent in linear woody habitats at arable farms (Schirmel *et al.*, 2016) but the scrub at Canvey Wick only recently closed canopy, so this comparison is problematic. Lower numbers of predatory carabids compared with disturbed areas may reflect the preference of some species for openness at ground level (Harvey *et al.*, 2008), or association with dynamically variable food sources at the landscape scale (Woodcock *et al.*, 2010).

For dispersal traits, macropterous species were associated with the phase one restoration. The same trait has been predominant after hydrological (Lambeets *et al.*, 2008; Bettacchioli *et al.*, 2012) and ground disturbances (Pedley and Dolman, 2014). Wing dimorphic species were associated with the phase two restoration. Dispersal dimorphism offers an advantage with macropterous individuals adapted for dispersal, and greater reproductive fitness of micropterous morphs (Roff, 1986). No increase of wing dimorphism was observed along an experimental ground disturbance gradient, which was attributed to a limited pool of potential colonists (Pedley and Dolman, 2014). Brachypterous species were positively associated with reference plots, where there was ground disturbance from rabbits. Wingless carabids are usually associated with undisturbed habitats (Ribera *et al.*, 2001), although the pattern

in the present study largely resulted from abundance of *Syntomus foveatus*, a xeric-associated species (Brändle *et al.*, 2000). Spring breeding carabids were less frequent in control plots. Compared with restored areas, control and reference plots were dry in the winter and less attractive to species breeding in substrate exposed by receding spring water levels (Lott, 2001).

Carabids with conservation status were particularly associated with plots a year after disturbance. There was a similar lag in response after experimental scrapes in a forest trackway network (Pedley *et al.*, 2013). Generalist species were less frequent in control plots compared with all other areas, where eurytopic and pioneer species were more abundant. High intensity flood disturbance similarly favoured colonisation by generalist carabids (Fournier *et al.*, 2015). Other research has linked smaller body size to disturbance (Ribera *et al.*, 2001) but it was not significant in the present study. This was probably a result of edge effects, as the restored areas were part of a heterogeneous habitat mosaic. Although habitat boundaries can strongly affect invertebrate activity (Bertoncelj and Dolman, 2013), the distance from undisturbed habitat (10 m) was small enough for results to be influenced by terrestrial movement. Pitfall traps were also found to over-sample large, ground-active predatory carabids, in comparison with smaller species (Andersen and Arneberg, 2016).

3.4.2 Staphylinid response traits

Smaller staphylinids were positively associated with the most recently disturbed plots. Trait-based studies of complete staphylinid assemblages are rare, so there is little context for this finding. At limestone grassland patches in a blanket peat landscape, smaller staphylinids were more frequent in aerial traps (Bauer, 1989). Larger staphylinids were more abundant at low levels of flood disturbance (Fournier *et al.*, 2015) and were associated with scattered shrubs rather than grassland in wood pasture (García-Tejero and Taboada, 2016). There is a performance advantage for predators of large invertebrates by energy maximisation (Kaspari and Joern, 1993) which may partly explain the lower frequency of larger staphylinids in extensive areas of bare ground. This is a prediction of the ‘enemy-free space’ hypothesis, where prey species have more chance of escaping in dense vegetation (Lawton, 1983). Body size could have other functional links, such as to foraging capability (Fountain-Jones *et al.*, 2015), although there is insufficient ecological information to explore this.

Species with conservation status were particularly associated with phase two plots, largely resulting from abundance of *Neobisnius procerulus*, which is typical of exposed secondary sites in the UK, such as gravel pits (Denton, 2002).

Disturbance was also a likely cause of trait shifts through resource availability. Species associated with dung and carrion were less frequent at control plots, and more frequent in the phase two restoration. This can probably be explained by rabbit activity, with dung concentrated in the most recently disturbed areas and not beneath trees and scrub. Similar effects were observed at exposed riverine sediments open to cattle and sheep (Eyre *et al.*, 2001) and grasslands in wood pasture (García-Tejero and Taboada, 2016). Staphylinid assemblages responded strongly to leaf litter accumulation at undisturbed plots, which has been identified as a critical resource for this family (McKenna *et al.*, 2015). Litter-associated species were less frequent at reference areas compared with restoration plots, despite deeper litter. This was probably linked to moisture levels (Hoffmann *et al.*, 2016), for which trait data were lacking. Mycetophilous species were associated with undisturbed plots, where decomposing organic matter and woody debris were likely to favour the development of fungi (Nordén *et al.*, 2004). The phytophagous trait was associated with plots a year after disturbance. Populations of algae-feeding *Bledius* and *Carpelimus* (Herman, 1986; De Marzo, 2016) probably increased with time after restoration, as phases one and two were both inundated by groundwater in winter.

3.4.3 Spider response traits

Spiders were larger at undisturbed plots, with smaller linyphiids more than five times as abundant in restored areas. The presence of smaller species in intensively disturbed habitats was predictable from the association of linyphiids with arable fields (Downie *et al.*, 2000) and land reclamation (Meijer, 1977). This also accounted for the positive association of web-hunting and ballooning species in both restoration areas, and the negative association with reference and control plots. Ballooning was associated with fire disturbance (Langlands *et al.*, 2011) but not scraping to remove vegetation (Pedley and Dolman, 2014). As ballooning is a passive dispersal mechanism, landscape permeability is probably important (Schmidt and Tscharntke, 2005). Other studies of hydrological (Lambeets *et al.*, 2008) and ground disturbance (Pedley and

Dolman, 2014) have reported larger spiders in disturbed areas. This has been interpreted as an adaptation for local terrestrial dispersal (Pedley and Dolman, 2014), although with other potential functional links including improved desiccation resistance in open habitats subject to increased thermal fluctuation (Langlands *et al.*, 2011).

Spider prey capture strategies are adaptations to habitat structure and resource availability. There was a significant association with running hunters a year after disturbance. Other studies have linked disturbance to an increase in cursorial hunters (Lambeets *et al.*, 2008; Pedley and Dolman, 2014) and sparse dune vegetation was favoured by spiders with higher levels of cursorial mobility (Bonte *et al.*, 2003a). Ambush hunting spiders were particularly associated with plots a year after disturbance, but not recent scrapes. Most ambush hunting spiders in this study exploited the litter layer (Jiménez-Valverde and Lobo, 2007) which was substantially reduced by scraping (Table 2.1, Chapter 2). Recovery of spider populations was also slow after removal of vegetation and soil organic matter in a grassland forest trackway network (Pedley *et al.*, 2013), which emphasises the need for longer term studies. Interpretations of spider hunting guilds in the present research were limited by reliance on pitfall trap sampling. Spiders with aerial webs were underrepresented in pitfall catches (Topping and Sunderland, 1992) and it was not possible to separate effects on this guild compared with sheet web predators (Uetz *et al.*, 1999).

The lower moisture tolerance for species outside phase one plots was probably caused by drier conditions under the canopy of control plots, and among very short vegetation in phase two and reference plots. Short, sparser vegetation was also a likely reason for the negative association of shade tolerant species with phase two and reference plots, compared with the positive association under closed canopy scrub. Vegetation succession had a fundamental effect structuring spider assemblages in other sand ecosystems (Buchholz *et al.*, 2010; Schirmel and Buchholz, 2011). This was probably partly related to microclimate, which was more extreme in early successional stages at sand dunes (Mantilla-Contreras *et al.*, 2011). Moisture and shade were also important drivers for beta-diversity of European spiders, especially in open habitats (Entling *et al.*, 2007). The positive association of species without conservation status in restoration areas, and negative association with control and reference

plots, reflected increased abundance of agrobiont and eurytopic spiders after intensive disturbance (Samu and Szinetár, 2002).

3.4.4 Functional indices

The overall pattern was for higher values for functional dispersion (F_{Dis}), functional divergence (F_{Div}) and functional evenness (F_{Eve}) at undisturbed plots. The extent to which these metrics explain different ecological attributes for communities is not entirely clear. Lower values for F_{Dis} potentially reflected limited niche availability, as in early successional stages at sand dunes (Schirmel *et al.*, 2012). Lower values for F_{Div} suggested that in the community assembly process after disturbance, many species shared traits, in agreement with previous studies of river floodplains (Fournier *et al.*, 2015). Similarly, lower values for F_{Eve} could result from a small number of abundant species that were functionally distant from the assemblage-level centroid of trait composition (Gerisch *et al.*, 2012). In contrast to these findings, carabid functional richness was higher at more disturbed sites within a gradient of agricultural and marginal land (Pakeman, 2014). This was partly explained by interactions with plant traits and productivity, and the difficulty of comparing assembly processes in a landscape with a long history of cultivation, to successional sequences where niches appear to increase in number as the vegetation develops.

Functional metrics have been linked to impacts on ecosystem function and stability. High functional divergence may represent increased ecosystem function, resulting from more efficient resource use (Mason *et al.*, 2005). Management to increase functional diversity has been recommended for carabids in conifer plantations (Spake *et al.*, 2016) and to enhance populations of predatory invertebrates and pollinators in arable landscapes (Woodcock *et al.*, 2014). These measures may not be applicable in all situations. Functional diversity was relatively stable for benthic macroinvertebrates in European rivers, even with severe land use impacts, whereas individual trait values measurably responded to environmental gradients (Vandewalle *et al.*, 2010). From the perspective of conservation management at Canvey Wick, there was a lack of congruence between functional indices and species conservation status, particularly for carabids and staphylinids. This suggests that functional metrics might not be appropriate for all site management objectives. The extent to which this is a priority depends on objectives for individual sites or landscapes.

Sites should be managed for their strengths, which depends on representation of particular features within the regional habitat pool (Kirby, 1992).

3.4.5 Generalisation between taxonomic groups

There was limited scope to generalise between invertebrate groups because of the lack of comparable trait information. Disturbance tended to filter less mobile species. Carabids with fully developed wings were more frequent, as well as spiders that dispersed by ballooning. Staphylinids were smaller, which may be linked to increased propensity for flight (Bauer, 1989). Shifts in resource preference were apparent for each group, although these were not directly comparable. Interpretation must be qualified, as while some traits may influence sensitivity to disturbance, outcome at the community level is determined by interactions with other species, environmental parameters, and the surrounding landscape (McIntyre *et al.*, 1999; Winqvist *et al.*, 2014). This relates to the wider difficulty in linking an assembly process to a specific spatial scale (Taudiere and Violle, 2015). The present study also only considered a single site, and without samples for a full year, missing the activity periods of some species.

Body size was the only common functional trait, with smaller staphylinids particularly associated with disturbance. No link was detected for carabids, and spiders were relatively larger in wooded areas. Lack of congruence is a recurrent theme in trait-based studies (Pedley and Dolman, 2014). This reflects the range of evolutionary responses that different species can develop under the same environmental conditions (Southwood, 1988). The effects of experimental design and landscape setting could also be important, if responses are on a larger scale than the environmental gradient investigated (Gerisch *et al.*, 2012) There is further difficulty in generalising trait responses, as body size is potentially linked to multiple ecological functions. In a study of community assembly after fire, larger spiders were more frequent in the early stages of succession (Langlands *et al.*, 2011). It was not possible to distinguish whether this was a morphological/physiological trait for desiccation resistance, or a functional adaptation for burrowing (Langlands *et al.*, 2011). This can complicate interpretation within a single group, as hydrological disturbance favoured larger linyphiid and smaller lycosid spiders (Lambeets *et al.*, 2008).

There is a clear need for common, robust foundations in functional trait research (Fountain-Jones *et al.*, 2015). Comparison of invertebrate studies is rarely straightforward, due to the lack of consistent methodology (Cornelissen *et al.*, 2003). As an example, records of ballooning propensity taken from different sources, or scored at genus rather than species level, are not directly comparable. Sampling methods may also have an effect, such as the greater probability of capturing large, cursorial spiders in pitfall traps where vegetation resistance is lower (Topping and Sunderland, 1992). The simplistic ecological data available for some behavioural traits could have limited the capacity to distinguish between stages of post-disturbance succession (Langlands *et al.*, 2011). The feeding habits of predatory staphylinids are poorly known and while many are generalist predators of soil arthropods (Bohac, 1999), variation in mandible morphology suggests that some species vary markedly in function. A broader range of morphological measurements might facilitate comparisons between groups (Fountain-Jones *et al.*, 2015), although this is still limited by the lack of genetic information to incorporate species-level phylogenetic correlations (Barton *et al.*, 2011).

Chapter 4 – Canvey Wick ground disturbance experiment

4.1 Introduction

Ecological succession to scrub and secondary woodland is well documented at brownfield sites (Shaw, 1992; Řehounková and Prach, 2006; Schadek *et al.*, 2009). There may also be alternative stable states of vegetation development (Prach and Pyšek, 2001). Tall graminoids such as *Elytrigia atherica* can form dense monospecific stands, as well as tussock forming species including *Festuca arundinacea* and *Dactylis glomerata* (Bakker and Berendse, 1999). High coverage of graminoids can limit colonisation by shrub and tree seedlings due to lack of niches for germination, reduced light availability at ground level, and competition for nutrients and water (Mantilla-Contreras *et al.*, 2011). Similar successional processes have been documented in semi-natural habitats including chalk grassland (Bobbink and Willems, 1987), heathland (Mantilla-Contreras *et al.*, 2011), dry coastal dunes (ten Harkel and van der Meulen, 1996) and dune slacks (Lammerts and Grootjans, 1997). Site-specific conditions determine the dominant graminoids in sandy ecosystems, including *Ammophila arenaria* and *Carex arenaria* at lime- and iron-poor sites (Kooijman and de Haan, 1995; Remke *et al.*, 2009), and *Calamagrostis epigejos*, *Elymus repens* and *Elytrigia atherica* on more calcareous substrates (Kooijman and van der Meulen, 1996).

Graminoid dominance may have multiple causes. Atmospheric deposition of nitrogen has been linked to changes in vegetation composition, and the growth response of different species (Field *et al.*, 2014). UK emissions of reduced and oxidised nitrogen have declined in recent decades (between -24% and -46%; 1990-2007) but deposition levels remain high due to changes in atmospheric chemistry (RoTAP, 2012). Phosphorous limitation might change the competitive balance in vegetation at high levels of nitrogen availability, which has been suggested as a cause of *Molinia caerulea* expansion at heathlands (Härdtle *et al.*, 2006). Other mechanisms could maintain or consolidate graminoid dominance. Nitrogen mineralisation was higher in grass-dominated plots (Veer and Kooijman, 1997) and there are probably positive feedbacks with nitrogen cycling (Wedin and Tilman, 1990). Disturbance dynamics were reduced after limiting public access (Remke and Blindow, 2011), and by the effect of myxomatosis on rabbits (Thomas, 1963; Isermann *et al.*, 2010). Some factors

may be important in combination, as grass encroachment increased in coastal dunes with high nitrogen deposition two years after grazing was excluded (ten Harkel and van der Meulen, 1996).

Grass encroachment affects plant species composition, patterns of succession, and environmental conditions (Mantilla-Contreras *et al.*, 2011). Species richness and diversity were twice as high in open dune grassland, compared with plots dominated by tall graminoids (Veer and Kooijman, 1997). Dune slacks were characterised by therophytes and hemicryptophytes in the early stages of succession (Olff *et al.*, 1993). Small, short-lived species decreased as rhizomatous monocots established, initially saltmarsh rush *Juncus gerardii* and later *Phragmites australis*. Lichens may also be affected, with a steady reduction in the species-rich *Violo-Corynephoretum* community in middle dune grasslands as the area dominated by *Ammophila arenaria* increased (Ketner-Oostra and Sýkora, 2004). In addition to plant communities, graminoid succession affects environmental conditions. Total above ground biomass in dunes was almost 10 times as high in plots dominated by tall graminoids, compared with open grassland (Veer and Kooijman, 1997). Light penetration in plots dominated by the grasses *Elymus athericus* and *Calamagrostis epigejos* was less than 10% of the level recorded in open vegetation (Kooijman and van der Meulen, 1996). The amount of ectorganic matter in open dune grassland plots was virtually nil, compared with 260 g m^2 in grass-dominated plots (Veer and Kooijman, 1997).

Invertebrate communities are also affected by increased graminoid dominance. Specialist early successional carabids were significantly more abundant at disturbed plots within a forest trackway network, compared with unmanaged grassland rides (Pedley *et al.*, 2013). Grass encroachment was linked to a reduction in dune spider assemblages associated with short grassland and humid valleys (Bonte *et al.*, 2002). Total spider abundance was greater in salt marshes invaded by the grass *Elytrigia atherica*, although coastal specialist species declined (Pétillon *et al.*, 2005). Other studies have identified conservation value in areas dominated by tall graminoids. Terrestrial invertebrate abundance and diversity was greatest on ungrazed marshes, particularly detritivores within a food web structured by tall vegetation and deep litter (Ford *et al.*, 2013). Orthoptera species richness was highest in grass-

invaded heath, including graminivorous Acrididae, chortobiont and thamnobiont species (Schirmel *et al.*, 2011). Grazed and ungrazed inland salt meadows contained distinct spider assemblages, including a high percentage of rare species (Zulka *et al.*, 1997). Fly ash deposits overgrown by *Calamagrostis epigejos* and other grasses were also important for spiders of conservation value (Tropek *et al.*, 2014).

Ground disturbance is a common management technique to restore invertebrate conservation interest associated with early successional habitats. In sandy calcareous grassland, there were significantly more red-listed carabids at ploughed plots than grazed plots (Olsson *et al.*, 2014). The value of bare ground creation for heathland carabids was investigated at patches between 1 m^2 and 100 m^2 (Cameron and Leather, 2012a). Species richness, abundance and diversity were highest on intermediate patches, although stone cover and surrounding grass tussocks were also important. The effects of deep and shallow perturbations were compared in sandy calcareous grassland, with distinct communities of Coleoptera and vascular plants associated with shallow disturbance (Ödman *et al.*, 2011). To assess persistence of created bare ground, swiping, harvesting, discing, forest ploughing, agricultural ploughing, and turf stripping were compared on forest ride verges (Pedley *et al.*, 2013). Effects of turf stripping and agricultural ploughing lasted longer and produced a similar carabid assemblage to heath reference sites. In lowland heathland, a distinct community of sand-associated carabids developed after sod-cutting to mineral soil (Schirmel, 2010). Less intensive ‘choppering’ left a thin layer of organic matter, and the carabid assemblage mainly comprised eurytopic species.

Current evidence of the benefit of ground disturbance for invertebrates comes from dry heathland and grasslands. Similar outcomes can be anticipated in seasonally wet grassland, but direct transfer of results cannot be assumed. Evidence is required, particularly as varied outcomes have been reported for less intensive disturbance techniques (Schirmel, 2010; Pedley *et al.*, 2013). The practical application of ground disturbance is rarely considered, but scraping and sod cutting produce large quantities of arisings (Niemeyer *et al.*, 2007). At one heathland, turves were taken away to a golf course, to avoid leaving mounds of material that could hinder future management (Lyons and Pryor,

2014). This is unlikely to be an option in most situations and a disturbance technique that does not generate arisings might be preferred (Britton *et al.*, 2000). More generally, at some sites disturbance management could be implemented in different habitats. It is difficult and expensive to reset succession where scrub is well-established (Mortimer *et al.*, 2000). Detailed information about invertebrate assemblages should inform decisions to adopt this approach, or reintroduce disturbance dynamics in areas dominated by tall graminoids.

The outcomes of ground disturbance are usually assessed for one or two invertebrate groups, commonly carabids and spiders. The effects on many other families and orders are poorly understood. The taxonomic scope in this study was broader, including all sampled spiders and Coleoptera. To provide insights into disturbance effects, both community composition and species functional traits were considered. Among European Coleoptera, most trait information is available for carabids. This has encouraged research linking functional traits to community assembly, including responses to landscape intensification (Ribera *et al.*, 2001), flood disturbance (Lambeets *et al.*, 2009) and forest management (Spake *et al.*, 2016). Functional traits have also been used to predict ecosystem functions, such as seedbank regulation (Bohan *et al.*, 2011). Other studies have tested trait responses for multiple taxonomic groups, as part of the search for ecological generality (Pedley and Dolman, 2014). Results are inconsistent, which probably reflects differing responses to disturbance and environmental resource factors (Lavorel and Garnier, 2002). A limitation for wider comparisons is the lack of identical trait sets in literature, apart from body size (Fountain-Jones *et al.*, 2015). In the present research, emergent groups were characterised from morphological traits measured for both carabids and staphylinids.

This study examines management techniques to benefit stenotopic invertebrate communities on a low-nutrient substrate, in an area dominated by perennial graminoids. The development of invertebrate communities was monitored before and after application of treatments with different disturbance intensity in November 2015. It was anticipated that the treatments would affect beetle and spider communities. To explore this, the following questions were asked: (1) Does the intensity of disturbance influence occurrence of stenotopic early

successional beetles or spiders, and generalist species? (2) Is it possible to identify indicator species for the different stages of succession? (3) How do the treatments affect functional group structure of beetles and spiders? (4) Were distinct carabid and staphylinid groups associated with the different treatments, based on morphological traits? (5) How does habitat structure develop in the year following disturbance?

4.2 Methods

4.2.1 Experimental design

The location for this research was Canvey Wick in Essex (51°31'19"N 0°32'38"E), a post-industrial site created in the early 1970s by disposal of marine dredgings (Figure 1.2). By 2015, mosaics of bare ground and pioneer vegetation had been replaced by perennial graminoids in some areas. To investigate potential management for stenotopic early successional invertebrates, 0.64 ha of species-poor grassland was divided into twenty-seven 100 m^2 experimental plots (Figure 4.1). These were monitored pre-treatment, between May and September 2015. Two ground disturbance techniques were applied in November 2015, from a 360-degree tracked excavator and bucket attachment. Disturbance was similar to rotovation (Britton *et al.*, 2000), although with vegetation and humus layers shallowly buried, rather than mixed with sub-soil (Figure 4.2). Scraping removed all vegetation and organic soil horizons, which were taken away from the experimental area on a dumper truck (Figure 4.3). Plots were in adjoining blocks of three, which was unavoidable because of the limited area available for experimentation. The order of application for treatments and controls was determined at random for each block. An unforeseen consequence was that two pairs of scraped plots, two pairs of control plots, and one pair of disturbed plots were adjacent. As a result these were larger, although sampled separately and treated as replicates in the analysis below. It would have been preferable to randomise the order of treatments in one block, and offset this throughout the experimental area.

4.2.2 Invertebrate sampling

Invertebrates were captured in three pitfall traps in each monitoring plot. These were arranged in a straight line, with adjacent traps separated by 1 m, and the middle trap placed in the centre of the plot. Traps were polypropylene cups, 8.0

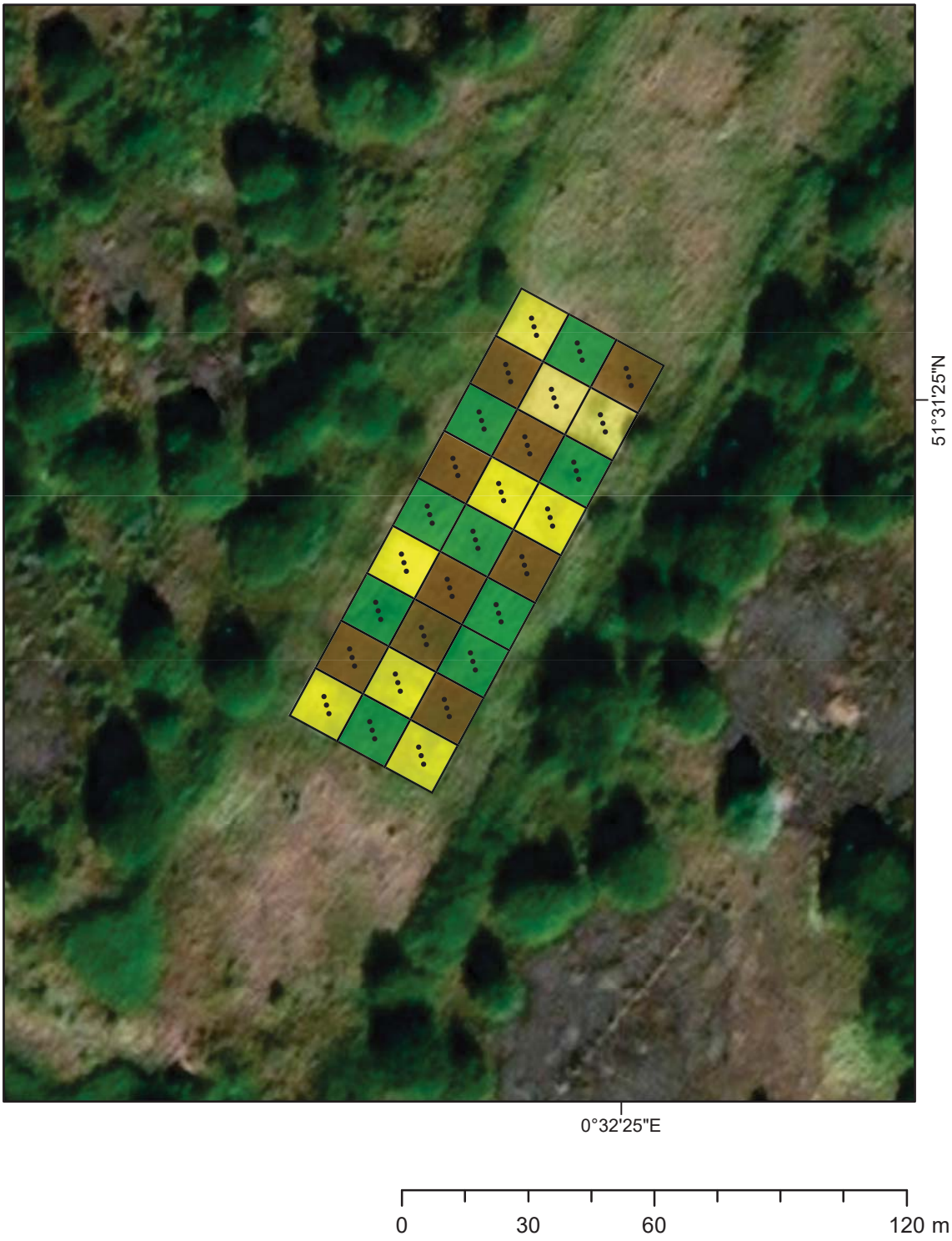

Figure 4.1 Experimental design for ground disturbance plots at Canvey Wick, Essex. Scraping removed all vegetation and organic soil horizons, which were taken away from the experimental area. Vegetation and humus layers were shallowly buried during excavator disturbance. Treatments were applied in November 2015. Basemap source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS user community.

Figure 4.2 Ground disturbance applied by 360-degree excavator with bucket attachment, in species-poor grassland at Canvey Wick, November 2015. Vegetation and humus layers were shallowly buried, exposing mineral soil.

Figure 4.3 Scraping applied by 360-degree excavator with bucket attachment, in species-poor grassland at Canvey Wick, November 2015. Vegetation and soil organic layers were removed from the experimental area.

cm diameter and 10.5 cm deep, with 50 ml preservative (50% propylene glycol and 50% distilled water) and a drop of detergent to break the surface tension. To reduce capture of small mammals and reptiles, all traps were covered with 13 mm x 13 mm wire mesh. Traps were open for sixteen-day periods in May, July and September, in 2015 and 2016. This sampling programme provided adequate seasonal coverage, including key periods for invertebrate activity in early successional habitats (Lott, 1998), and minimised potential depletion effects (Digweed *et al.*, 1995). Samples were preserved in 70% industrial methylated spirits.

4.2.3 Invertebrate classification

Adult Coleoptera were determined to species, and classified as stenotopic in early successional habitats, generalists, or eurytopic, from the references in Table 4.1. Adult spiders were determined to species from Roberts (1985) and Roberts (1987), and classified as stenotopic in early successional habitats, generalists, or eurytopic from Harvey *et al.* (2002). Some specimens could not be identified from morphological characteristics, or genitalia. Staphylinids were female *Gabrius*, *Ochthephilum collare*/*fracticornis*, *Carpelimus bilineatus*/*erichsoni*, *Carpelimus rivularis/similis*, *Anotylus sculpturatus/mutator*, *Gyrohypnus angustatus/wagneri*, and males and females of the *Mocyta fungi* complex. Spiders were female *Dicymbium nigrum*/*brevisetosum*, and all immature specimens, which lacked developed reproductive structures. All invertebrates were also classified in functional groups based on adult behaviour, as predatory, zoophagous (predatory and scavenging), phytophagous (herbivorous, or at least primarily granivorous), detritivorous (feeding on detritus, including saprophagous, mycetophagous and coprophagous species), or unknown. Information was taken from the references in Table 4.1, and additionally Cooter and Barclay (2006), Anderson *et al.* (2013), Ford *et al.* (2013), and Talarico *et al.* (2016).

4.2.4 Plant species composition and habitat structure

Data on plant species composition and habitat structure were collected in 2015 and 2016, in June. Within a 1 m x 4 m area at each side of the pitfall trap row, vegetation was surveyed in four 50 cm x 50 cm sub-units. The quadrats were randomised at each survey visit, because of the potential to change the

Table 4.1 Identification references for Coleoptera.

Family	Reference
Anthicidae	Telnov (2010)
Apionidae	Duff (2016)
Byrrhidae	Joy (1932)
Cantharidae	Fitton (1973)
Carabidae	Luff (2007), Duff (2012)
Cerambycidae	Duff (2016)
Chrysomelidae	Duff (2016)
Clambidae	Johnson (1966)
Coccinellidae	Majerus *et al.* (1989)
Corylophidae	Bowstead (1999)
Cryptophagidae	Coombs and Woodroffe (1955), Lohse and Klausnitzer (1992)
Curculionidae	Duff (2016)
Dytiscidae	Foster and Friday (2011)
Elateridae	Joy (1932)
Erirhinidae	Duff (2016)
Erotylidae	Joy (1932)
Helophoridae	Foster *et al.* (2014)
Histeridae	Duff (2012)
Hydraenidae	Duff (2012)
Hydrophilidae	Foster *et al.* (2014)
Kateretidae	Kirk-Spriggs (1996)
Latridiidae	Freude *et al.* (1967)
Leiodidae	Cooter (1996)
Malachiidae	Joy (1932)
Nitidulidae	Audisio (1993)
Oedemeridae	Buck (1954)
Ptiliidae	Duff (2012)
Scarabaeidae	Jessop (1986)
Silphidae	Duff (2012)
Staphylinidae	Hammond (1972), Freude *et al.* (1974), Lott (2009), Lott and Anderson (2011), Assing and Schülke (2012)
Tenebrionidae	Brendell (1975)
Throscidae	Lucht and Klausnitzer (1998)

development of vegetation by disturbance during repeated monitoring. In each quadrat, percentage cover of vascular plant species were estimated, with identifications from Stace (2010). Vegetation height was measured at four random positions, with a 1 m measuring pole and 10 cm cross beam, lowered until touching vegetation. Measurements of sward height by sward stick, drop disc and direct measurement were consistent for each method but not interchangeable (Stewart *et al.*, 2001). The method in the present study was preferred because of the very short plants at scraped plots (Woodcock *et al.*, 2010). Mean vegetation height was calculated for each quadrat from the four height measurements. Percentage cover of moss and bare ground were estimated, and litter depth was measured to the nearest millimetre.

4.2.5 Statistical analysis

Univariate analyses were completed in the statistical software R (R Core Team, 2017). Multivariate analyses were completed with the Canoco 5.0 software package (ter Braak and Šmilauer, 2014). Precise *p*-values are reported, to allow interpretation of *p* as a measure of evidence, and to facilitate meta-analysis (Cumming and Calin-Jageman, 2017).

4.2.5.1 Species accumulation curves

To assess whether sampling effectively captured assemblage composition for each taxonomic group, rarefaction curves were constructed (Chao *et al.*, 2014) using the iNext package (Hsieh *et al.*, 2016).

4.2.5.2 Generalized linear mixed models

Abundance and species richness of stenotopic and generalist species were compared between treatments with generalized linear mixed models (GLMMs). This approach was appropriate for count data with a non-Gaussian distribution, zero probability of negative values, and heterogeneity in the mean variance relationship (Zuur *et al.*, 2009). It also allowed for potential correlation in the arrangement of treatments and controls as a randomised complete block design (Bolker *et al.*, 2009). The catches from individual traps within each monitoring plot were pooled, to produce single totals for each species per plot year. It was not possible to analyse individual traps or sampling periods, due to zero inflation in the subsets of data created by the species classification. Pearson residuals

from fitted models were calculated to check for overdispersion and determine the appropriate distribution for the response variable (normal, Poisson, or negative binomial). These were obtained by dividing each residual by the square root of the variance (Zuur *et al.*, 2009).

Residuals were plotted for model validation, to assess homogeneity versus fitted values, and versus the covariate. Spatial autocorrelation of residuals was checked by calculation of Moran's I in the ape package (Paradis *et al.*, 2004), based on a matrix of inverse Euclidean distance weights. Where this was significant ($P < 0.05$), an autocovariate was computed using the spdep package (Bivand *et al.*, 2013; Bivand and Piras, 2015), based on a distance-weighted function of neighbouring response values (Dormann *et al.*, 2007). This was included as an extra parameter in the model for the response variable. Differences among treatment means were examined by Tukey pair-wise comparisons. To adjust *p*-values, the family-wise error rate was controlled using the Holm-Šidák adjustment, as described in section 4.2.5.1 (Holm, 1979).

4.2.5.3 Environmental variables

GLMMs were not able to consistently predict the environmental variables across all treatment classes, due to heteroscedasticity. As a result, the variables were analysed with the Kruskal–Wallis (*H*) test (Kruskal and Wallis, 1952), followed by Dunn's test to report the results among multiple pairwise comparisons (Dunn, 1964). To adjust *p*-values, the family-wise error rate was controlled using the Holm-Šidák adjustment (Holm, 1979) in the dunn.test package (Dinno, 2016). This is a progressive step-up procedure assuming dependence between tests, which are completed in order of increasing probability $\alpha/n\text{-}1$. This was preferred to the classical Bonferroni correction, with tests compared to α/n, which gives a smaller or equal probability of rejecting false hypotheses for the same data (Holm, 1979).

4.2.5.4 Ordination

Assemblage composition for stenotopic beetles and spiders was examined with non-metric multidimensional scaling (NMDS). This was preferred due to the probability of considerable species turnover along the treatment gradient, which can cause a strong arch effect in metric ordination techniques (Prentice, 1977). Stenotopic species were compared at control plots, excavator disturbance plots,

and scraped plots. To produce a stable NMDS solution from the data available, samples were pooled within years for each experimental plot. Ordinations were completed with presence/absence data, and repeated with abundance data for alternative representation of the community structure. Ordinations were based on a matrix of Jaccard dissimilarities for presence/absence data, and Bray-Curtis dissimilarities for abundance data, with a maximum of 100 random starts to search for the lowest stress solution, k = 3. Stress values were compared with guidelines suggested by Clarke (1993): <0.05 = excellent, <0.10 = good, <0.20 = usable, >0.20 = not acceptable.

4.2.5.5 Indicator species

The association of species with treatments was assessed by indicator species analysis (Dufrene and Legendre, 1997). This was completed with the indicspecies 1.7.6 package (De Cáceres and Legendre, 2009) using the multipatt command (with func = ‘‘IndVal.g’’, indicator species of individual site groups only). This gives the conditional probability of species fidelity and specificity for a treatment. An index value is derived from these two quantities, with a permutation test (nperm = 999) for statistical significance. The index value is group-equalised proportional to the number of sites it contains. In this study, only species with an IndVal.g association index significant at alpha < 0.0001 are displayed. The tests are meaningful as the site groups are management treatments, rather than classified from species (De Cáceres, 2013).

4.2.6 Functional trait analysis

4.2.6.1 Morphological traits

Fifteen quantitative measurements and one qualitative character (Table 4.2) were taken from 40 species of carabid, and 44 species of staphylinid. These were species where a male and female could be recovered from the pitfall samples. This resulted in the exclusion of 20 carabid and 54 staphylinid species, recorded as either single individuals, or multiple individuals of the same sex. Measurements were taken with a micrometer in a stereomicroscope, at magnifications between 7.8x and 160x. While measurement of at least six specimens was recommended to capture most within-species variation (Fountain-Jones *et al.*, 2015), this level of replication was not possible due to

Table 4.2 Morphological variables for carabid and staphylinid species at Canvey Wick experimental ground disturbance plots. (a) Quantitative traits measured from specimens. (b) Qualitative trait from literature.

Code	Description
(a)	
HEW	Width of the head, measured behind the eyes
EYW	Diameter of the eye, measured from above
ANL	Length of the antenna
MAL	Length of the longest mandible
PRL	Medial length of pronotum
PRW	Maximum width of pronotum
ELL	Length of the elytra, from base of scutellum to apex of elytra
ELW	Maximum width of the elytra
TRL	Length of metatrochanter
HFL	Length of metafemur
HTL	Length of metatibia
HRL	Length of metatarsi
FFL	Length of profemur
FTL	Length of protibial
FRL	Length of protarsi
(b)	
WING	Wing development (apterous or brachypterous; dimorphic; macropterous)

time constraints. As a result, measurements might be atypical for a species, although it is likely that trait variation within a species is smaller than variation across all sampled species (Simons *et al.*, 2016). All morphological measurements were positively correlated with overall body length, so further variables were derived from these (Table 4.3). Data for derived variables are presented in Appendix 9.4, Table A9.4.1 (page 314). Fully resolved phylogenetic trees were not available for carabids and staphylinids, so no correction was made for potential phylogenetic correlations between traits among species (Pedley and Dolman, 2014). These may also contribute to the explanation of trait distributions (Chazdon *et al.*, 2003).

4.2.6.2 Indirect functional approach – emergent group analysis

Emergent groups were defined from separate biological trait matrices for carabid and staphylinid species. Cluster analysis was completed by agglomeration, with Ward's criterion to determine the decrease in variance for the cluster being merged (Ward, 1963). This approach was adopted as there were quantitative and qualitative traits, with the dissimilarity matrix calculated from Gower's coefficient (Gower, 1971). The number of emergent groups was determined by visual examination of a dendrogram representing the clustering solution. Statistical methods are available for this but have rarely been used because of the need for judgements about ecological validity in the definition of groups (Aubin *et al.*, 2009). The aim was to delimit relatively few groups through a synthesis of the morphological traits, based on subjective assessment that these contained ecologically relevant information (Aubin *et al.*, 2009). The relative occurrence of emergent groups for different treatments was analysed with generalized linear mixed models from a matrix of 2016 abundance data, following the procedure in Section 4.2.5.3. Species richness for emergent groups was not analysed, due to the small size of some emergent groups, within the limited subset of carabids and staphylinids selected for morphological characterisation.

4.3 Results

4.3.1 Environmental variables

Herbaceous plant species richness did not differ significantly between all treatments ($\chi^2_{[2, N=27]} = 5.73$, $P = 0.057$), although this was significantly higher at

Table 4.3 Morphological variables for carabid and staphylinid species at Canvey Wick experimental ground disturbance plots, derived from measured characters (Table 4.2).

Code	Description
LOG/PEL	Medial length of pronotum plus length of elytra, log(+1)-transformed
EYW/HEW	Diameter of the eye divided by the width of the head
ANL/PEL	Length of the antenna divided by LOG/PEL
MAL/PEL	Length of the longest mandible divided by LOG/PEL
PRL/PRW	Length of pronotum divided by width of pronotum
ELW/ELL	Maximum width of elytra divided by maximum length of elytra
TRL/HFL	Length of metatrochanter divided by length of metafemur
HFL/HLL	Length of metafemur divided by total length of hind leg
HTL/HLL	Length of metatibia divided by total length of hind leg
HRL/HLL	Length of metatarsi divided by total length of hind leg
FFL/FLL	Length of profemur divided by total length of front leg
FTL/FLL	Length of protibia divided by total length of front leg
FRL/FLL	Length of protarsi divided by total length of front leg

disturbed plots than scraped plots (P_{adj} = 0.0311) and marginally higher at disturbed plots than control plots (P_{adj} = 0.0869) (Table 4.4). Graminoid species richness differed significantly between treatments ($\chi^2_{[2, N=27]}$ = 17.05, P = 0.0002). This was significantly higher in control plots (P_{adj} = 0.0017) and disturbed plots (P_{adj} < 0.0001) than scraped plots. Shrub species richness differed significantly between treatments ($\chi^2_{[2, N=27]}$ = 14.09, P = 0.0008). This was significantly higher at control plots (P_{adj} = 0.0003) and disturbed plots (P_{adj} = 0.0194) than scraped plots. Mean field layer height differed significantly between treatments ($\chi^2_{[2, N=27]}$ = 14.09, P = 0.0008). This was significantly higher at control plots (P_{adj} < 0.0001) and disturbed plots (P_{adj} = 0.0142) than scraped plots, and significantly higher at control plots than disturbed plots (P_{adj} = 0.0103).

Moss cover differed significantly between treatments ($\chi^2_{[2, N=27]}$ = 16.84, P = 0.0002). This was significantly higher at control plots (P_{adj} = 0.0016) and disturbed plots (P_{adj} = 0.0002) than scraped plots. Litter depth differed significantly between treatments ($\chi^2_{[2, N=27]}$ = 24.02, P < 0.0001). This was significantly higher at control plots (P_{adj} < 0.0001) and disturbed plots (P_{adj} = 0.0071) than scraped plots, and significantly higher at control plots than disturbed plots (P_{adj} = 0.0142). Bare ground differed significantly between treatments ($\chi^2_{[2, N=27]}$ = 22.36, P < 0.0001). This was significantly higher at scraped plots than control plots (P_{adj} < 0.0001) or disturbed plots (P_{adj} = 0.0056), and significantly higher at disturbed plots than control plots (P_{adj} = 0.0264). An additional difference between treatments, although not quantified, was surface groundwater at the scraped plots between December 2015 and April 2016, with total inundation to a maximum depth of about 9 cm. The soil was saturated at control and disturbed plots, although with only scattered puddles of water at the surface (< 50 cm wide). Surface water was absent from the plots after April, in 2015 and 2016.

4.3.2 Invertebrate capture summary

During the study 25,021 invertebrates of 388 species were identified, comprising 16,813 beetles and 8,208 spiders. A further 265 invertebrates, probably representing six beetle and one spider species, could not be identified from female specimens. It is unlikely that this affected overall species richness

Table 4.4 Medians (minimum-maximum) and results of Kruskal–Wallis (H) test, comparing environmental characteristics among ground disturbance treatments at Canvey Wick in 2016. Treatments were excavator disturbance (n=9), scraping (n=9), and unmanaged grassland controls (n=9). Vegetation species were surveyed in June, and structural variables were compiled in August. Post-hoc tests were Tukey pairwise comparisons after Holm-Šidák adjustment, treatments with shared superscript a-c (ranked highest to lowest) did not differ significantly ($P > 0.05$).

Variable	Control	Disturbance	Scraping	Kruskal-Wallis $_{[2, N=27]}$
Herbaceous species richness	5 (2–10) ^{b}	8 (3–14) ^{a}	4 (3–6) ^{b}	$H = 5.73$
Graminoid species richness	7 (5–9) ^{a}	7 (6–9) ^{a}	4 (3–5) ^{b}	$H = 17.05$
Shrub species richness	2 (0–3) ^{a}	1 (0–2) ^{a}	0 (0–0) ^{b}	$H = 14.09$
Field layer height (cm)	26.3 (22.7–31.4) ^{a}	15.6 (10.5–22.8) ^{b}	1.6 (0.4–2.1) ^{c}	$H = 22.73$
Moss cover (%)	3.2 (0–21.5) ^{a}	6.0 (2.5–20.5) ^{a}	0 (0–0) ^{b}	$H = 16.84$
Litter depth (cm)	5.4 (4.1–6.7) ^{a}	0.6 (0.5–0.8) ^{b}	0 (0–0) ^{c}	$H = 24.02$
Bare ground (%)	0 (0–0) ^{c}	1.2 (0–8.5) ^{b}	89.1 (79.6–93.6) ^{a}	$H = 22.36$

statistics, as male specimens of all seven species expected to be present were identified. Seventeen male and female specimens in the staphylinid genus *Mocyta* were not identified because of taxonomic uncertainty. These, and female specimens for the other seven species, were excluded from subsequent analysis. A sample-based rarefaction curve estimated 77 (SE ± 7) spider species likely to be captured by pitfall traps in the sampled habitats in 2015 (Figure 4.4). Sixty-six species captured represent 85.7% of this estimated fauna. From an estimated 343 species of beetle (SE ± 56), 56.3% or 193 species were captured (Figure 4.4). A sample-based rarefaction curve estimated 96 (SE ± 13) spider species likely to be captured by pitfall traps in the sampled habitats in 2016 (Figure 4.5). Seventy-five species captured represent 78.1% of this estimated fauna. From an estimated 333 species of beetle (SE ± 25), 75.1% or 250 species were captured (Figure 4.5).

4.3.3 Beetle summary

Totals for beetles trapped at the monitoring plots were 9,205 in 2015 and 7,607 in 2016 (Appendix 9.4, Table A9.4.2, page 314). The number of individuals increased at the untreated control plots from 3,063 in 2015, to 3,153 in 2016. From 137 species trapped in 2015, the most abundant were *Amara lunicollis* (23%), *Drusilla canaliculata* (13%) and *Amara communis* (9%). There were 81 species represented by three or fewer individuals, forming 4.1% of the catch. From 160 species trapped in 2016, the most abundant were *Drusilla canaliculata* (17%), *Amara lunicollis* (10%) and *Amara communis* (7%). There were 97 species represented by three or fewer individuals, forming 4.5% of the catch.

The number of individuals decreased at control plots that were subsequently disturbed, from 3,068 in 2015, to 2,881 in 2016. From 132 species trapped in 2015, the most abundant were *Amara lunicollis* (23%), *Drusilla canaliculata* (18%) and *Amara communis* (7%). There were 77 species represented by three or fewer individuals, forming 3.7% of the catch. From 165 species trapped in 2016, the most abundant were *Aleochara brevipennis* (18%), *Agriotes lineatus* (8%) and *Pterostichus nigrita* (8%). There were 104 species represented by three or fewer individuals, forming 5.1% of the catch.

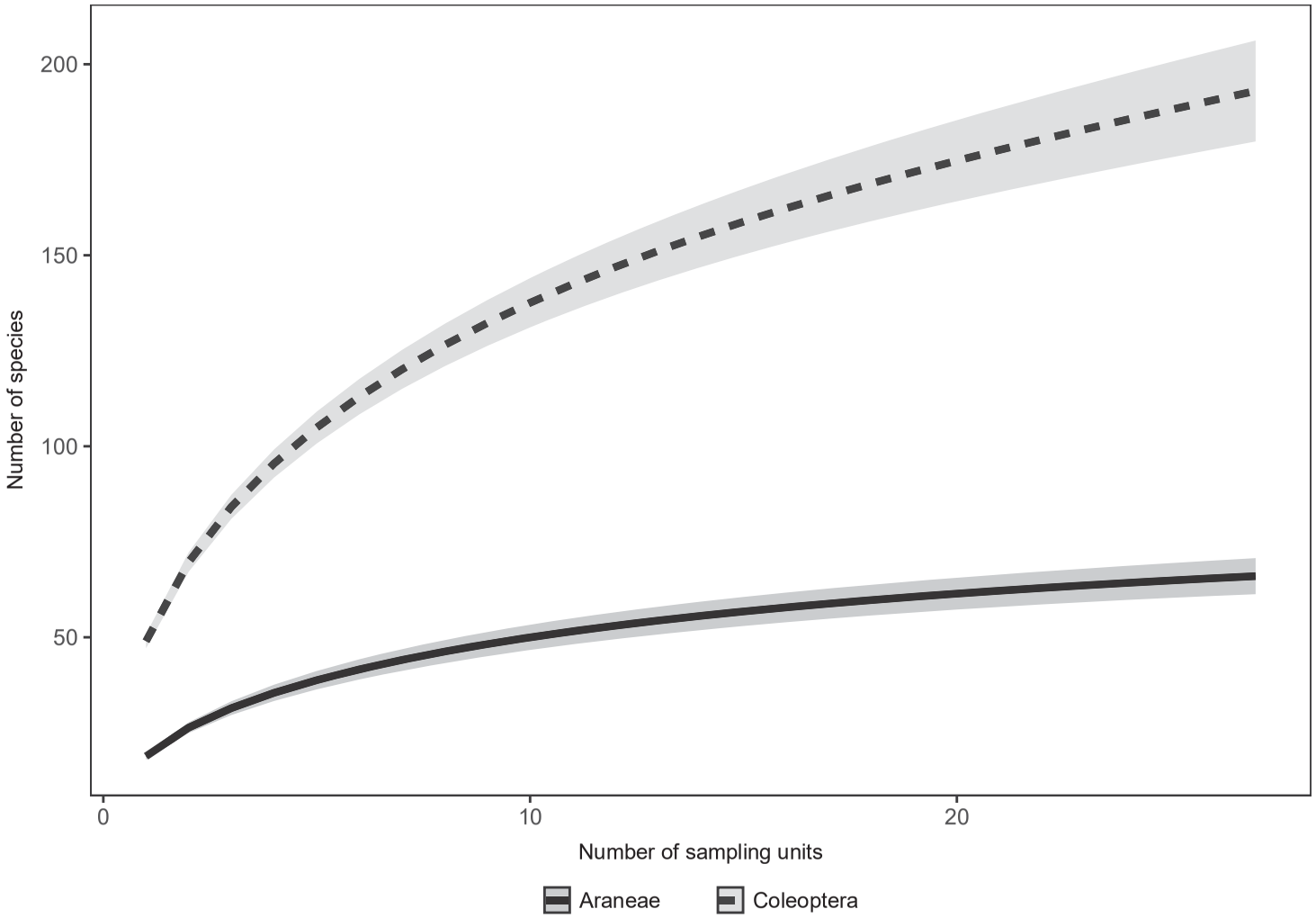

Figure 4.4 Sample-based rarefaction curves of species richness for beetles and spiders, based on species likely to be captured by pitfall traps in unmanaged grassland plots at Canvey Wick in 2015 (n=27). Shaded areas represent 95% confidence intervals. Curves become more shallow approaching the asymptotes of the estimated faunas.

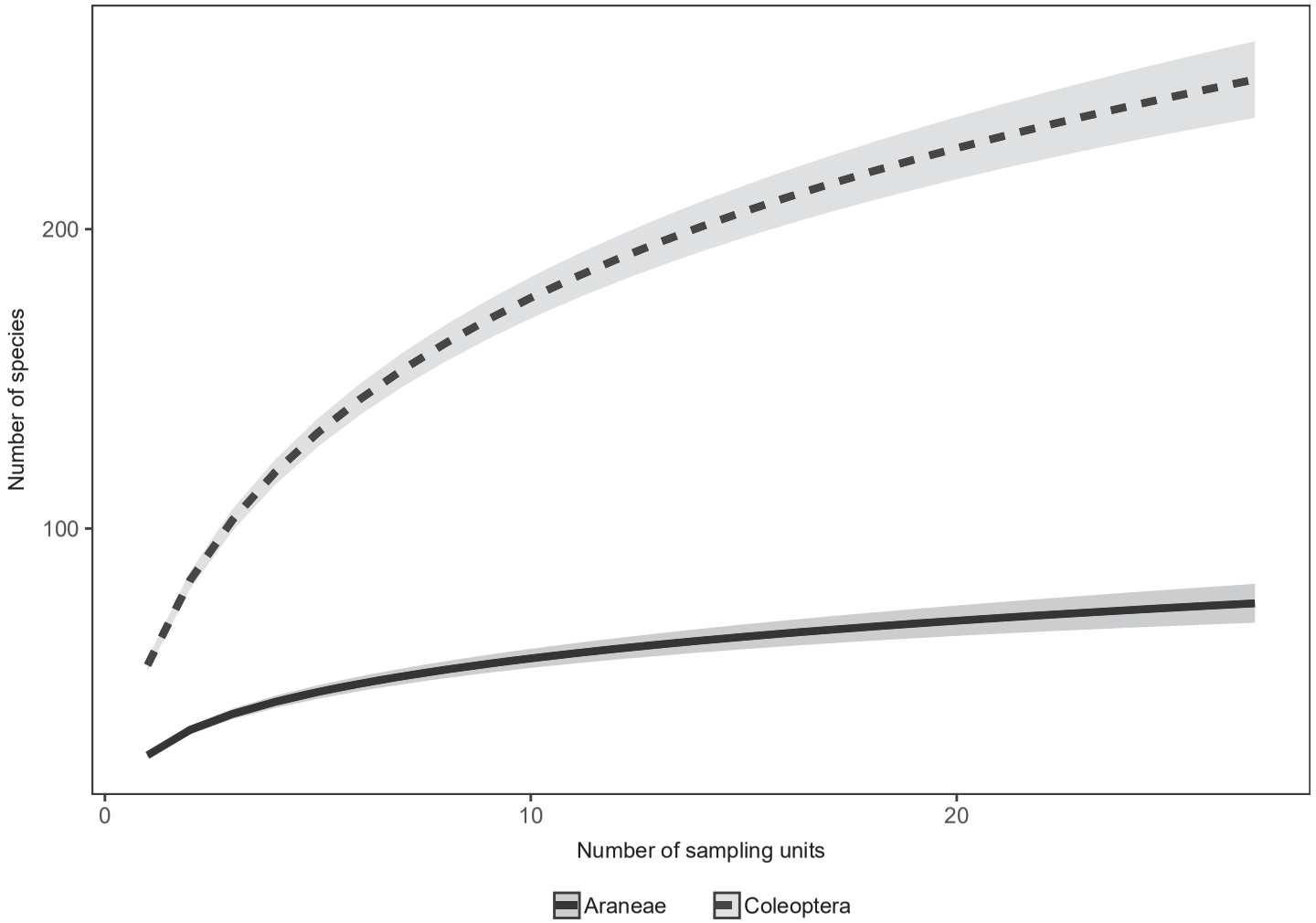

Figure 4.5 Sample-based rarefaction curves of species richness for beetles and spiders, based on species likely to be captured by pitfall traps in excavator disturbance, scraping, and unmanaged grassland plots at Canvey Wick in 2016 (n=27). Shaded areas represent 95% confidence intervals. Curves become more shallow approaching the asymptotes of the estimated faunas.

The number of individuals decreased at the control plots that were subsequently scraped, from 3,074 in 2015, to 1,573 in 2016. From 133 species trapped in 2015, the most abundant were *Amara lunicollis* (25%), *Drusilla canaliculata* (13%) and *Amara communis* (8%). There were 74 species represented by three or fewer individuals, forming 2.4% of the catch. From 156 species trapped in 2016, the most abundant were *Bledius gallicus* (9%), *Calodera protensa* (13%) and *Xantholinus longiventris* (8%). There were 100 species represented by three or fewer individuals, forming 9.4% of the catch.

4.3.4 Spider summary

Totals for spiders trapped at the monitoring plots were 3,876 in 2015 and 4,332 in 2016 (Appendix 9.4, Table A9.4.3, page 314). The number of individuals increased at the untreated control plots from 1,312 in 2015, to 1,583 in 2016. From 47 species trapped in 2015, the most abundant were *Pardosa pullata* (35%), *Alopecosa pulverulenta* (24%) and *Trochosa terricola* (10%). There were 25 species represented by three or fewer individuals, forming 2.8% of the catch. From 51 species trapped in 2016, the most abundant were *Hahnia nava* (22%), *Pardosa pullata* (21%), and *Alopecosa pulverulenta* (16%). There were 23 species represented by three or fewer individuals, forming 2.1% of the catch.

The number of individuals increased at the control plots that were subsequently disturbed, from 1,154 in 2015, to 1,840 in 2016. From 48 species trapped in 2015, the most abundant were *Alopecosa pulverulenta* (33%), *Pardosa pullata* (30%) and *Trochosa terricola* (9%). There were 28 species represented by three or fewer individuals, forming 3.8% of the catch. From 55 species trapped in 2016, the most abundant were *Pardosa pullata* (29%), *Alopecosa pulverulenta* (10%) and *Hahnia nava* (9%). There were 25 species represented by three or fewer individuals, forming 1.8% of the catch.

The number of individuals decreased at the control plots that were scraped, from 1,410 in 2015, to 909 in 2016. From 51 species trapped in 2015, the most abundant were *Pardosa pullata* (33%), *Alopecosa pulverulenta* (30%) and *Trochosa terricola* (9%). There were 27 species represented by three or fewer individuals, forming 2.8% of the catch. From 51 species trapped in 2016, the most abundant were *Erigone dentipalpis* (15%), *Erigone atra* (14%) and *Arctosa*

leopardus (12%). There were 23 species represented by three or fewer individuals, forming 3.7% of the catch.

4.3.5 Management community

4.3.5.1 Beetles

There was no significant difference in abundance ($\chi^2_{[2, N=27]} = 0.68$, $P = 0.7129$) or species richness ($\chi^2_{[2, N=27]} = 1.90$, $P = 0.3864$) of stenotopic beetles between treatments in 2015 (Table 4.5). Stenotopic beetles differed significantly in abundance ($\chi^2_{[2, N=27]} = 659.35$, $P < 0.0001$) and richness ($\chi^2_{[2, N=27]} = 46.89$, $P < 0.0001$) between treatments in 2016. Abundance and richness of stenotopic species were significantly higher at scraped plots than control or disturbed plots ($P_{adj} < 0.0001$ in all cases). There was no significant difference in abundance ($\chi^2_{[2, N=27]} = 0.27$, $P = 0.8717$) or species richness ($\chi^2_{[2, N=27]} = 0.77$, $P = 0.3864$) of generalist beetles between treatments in 2015. Generalist beetles differed significantly in abundance ($\chi^2_{[2, N=27]} = 59.94$, $P < 0.0001$) and richness ($\chi^2_{[2, N=27]} = 21.51$, $P < 0.0001$) between treatments in 2016. Abundance of generalist species was significantly higher at control and disturbed plots than scraped plots ($P_{adj} < 0.0001$ in both cases), and significantly higher at control plots than disturbed plots ($P_{adj} = 0.0097$). Richness of generalist species was significantly higher at control plots ($P_{adj} < 0.0001$) and disturbed plots ($P_{adj} = 0.0004$) than scraped plots.

NMDS based on presence/absence (stress value 0.1448) showed that stenotopic beetle communities were separated at scraped plots (Figure 4.6). Control and disturbed plots overlapped in both years, although some disturbed plots shifted towards the scraped community. Disturbed plots were narrowly separated by NMDS for abundance (stress value 0.1295), which was otherwise similar (Figure 4.7). Eleven species were unique to scrapes, with less than 10 individuals except for *Dyschirius politus* (16). At least 90% of captures were made at scraped plots for *Bledius gallicus* (143), *Carpelimus corticinus* (83), *Carpelimus gracilis* (79), *Elaphrus riparius* (70) and *Neobisnius procerulus* (45). Only *Dyschirius luedersi* and *Sitona cylindricollis* were unique to disturbed plots, both as single specimens. Eight species with conservation status were not part of the disturbance management community. Six species increased after the treatments, four with more than ten individuals. *Pterostichus gracilis* (50),

Table 4.5 Mean ± standard error for species richness and abundance of beetles and spiders at Canvey Wick ground disturbance experimental plots in 2015-2016. Generalized linear mixed models (χ^2, F) compare stenotopic species associated with early successional habitats, and generalists associated with grassland, woodland, or wetlands. Treatments were excavator disturbance (n=9), scraping (n=9), and unmanaged grassland controls (n=9). In each year, samples were pooled from three pitfall traps within each monitoring plot, open for 16 days in May, July, and September. Post-hoc tests show Tukey pairwise comparisons after Holm-Šidák adjustment, treatments with shared superscript a-c (ranked highest to lowest) did not differ significantly ($P > 0.05$). Moran's I was calculated to test for spatial correlation in GLMM residuals, based on a matrix of inverse Euclidean distance weights. Where this was significant ($P < 0.05$), an autocovariate was included in the model, calculated from a distance-weighted function of neighbouring response values (Dormann *et al.*, 2007).

Invertebrate group	Year	Classification	Measure	Control	Disturbance	Scraping	$GLMM_{[2, N=27]}$	Moran's I_{GLM}
Coleoptera	2015	Stenotopic	Abundance	3.9 ± 0.9 ^{a}	4.1 ± 0.7 ^{a}	4.7 ± 1.0 ^{a}	χ^2 = 0.68	0.3389
			Species richness	2.6 ± 0.4 ^{a}	3.0 ± 0.5 ^{a}	3.3 ± 0.6 ^{a}	*F* = 1.90	0.6017
		Generalist	Abundance	228.8 ± 28.5 ^{a}	217.7 ± 21.7 ^{a}	235.2 ± 21.5 ^{a}	χ^2 = 0.2745	0.6272
			Species richness	28.6 ± 1.8 ^{a}	26.7 ± 1.0 ^{a}	28.6 ± 1.9 ^{a}	*F* = 0.77	0.3050
	2016	Stenotopic	Abundance	4.7 ± 0.9 ^{b}	8.2 ± 0.8 ^{b}	58.9 ± 6.0 ^{a}	χ^2 = 183.62	0.2933
			Species richness	3.8 ± 0.7 ^{b}	5.4 ± 0.6 ^{b}	12.1 ± 0.4 ^{a}	*F* = 20.71	0.3481
		Generalist	Abundance	199.3 ± 27.3 ^{a}	134.0 ± 15.7 ^{b}	57.8 ± 7.2 ^{c}	χ^2 = 59.94	0.1840
			Species richness	30.4 ± 1.2 ^{a}	28.9 ± 0.8 ^{a}	20.2 ± 1.7 ^{b}	*F* = 21.51	0.6375
Araneae	2015	Stenotopic	Abundance	10.3 ± 1.4 ^{a}	7.2 ± 1.7 ^{a}	8.8 ± 1.5 ^{a}	χ^2 = 2.18	0.1912
			Species richness	3.6 ± 0.4 ^{a}	4.0 ± 0.7 ^{a}	4.9 ± 0.5 ^{a}	*F* = 1.97	0.7547
		Generalist	Abundance	58.9 ± 9.8 ^{a}	46.2 ± 5.4 ^{a}	58.8 ± 10.7 ^{a}	χ^2 = 1.19	0.0249
			Species richness	5.7 ± 0.6 ^{a}	5.8 ± 0.5 ^{a}	5.3 ± 0.6 ^{a}	*F* = 0.17	0.8287
	2016	Stenotopic	Abundance	26.0 ± 4.8 ^{a}	21.7 ± 3.4 ^{a}	7.1 ± 1.1 ^{b}	χ^2 = 32.81	0.7139
			Species richness	4.0 ± 0.4	4.8 ± 0.4	4.3 ± 0.4	*F* = 0.63	0.5445
		Generalist	Abundance	55.6 ± 9.3 ^{b}	68.1 ± 7.1 ^{a}	15.3 ± 3.1 ^{c}	χ^2 = 348.66	0.1521
			Species richness	6.2 ± 0.9 ^{a}	5.1 ± 0.6 ^{ab}	4.0 ± 0.6 ^{b}	*F* = 4.38	0.0200

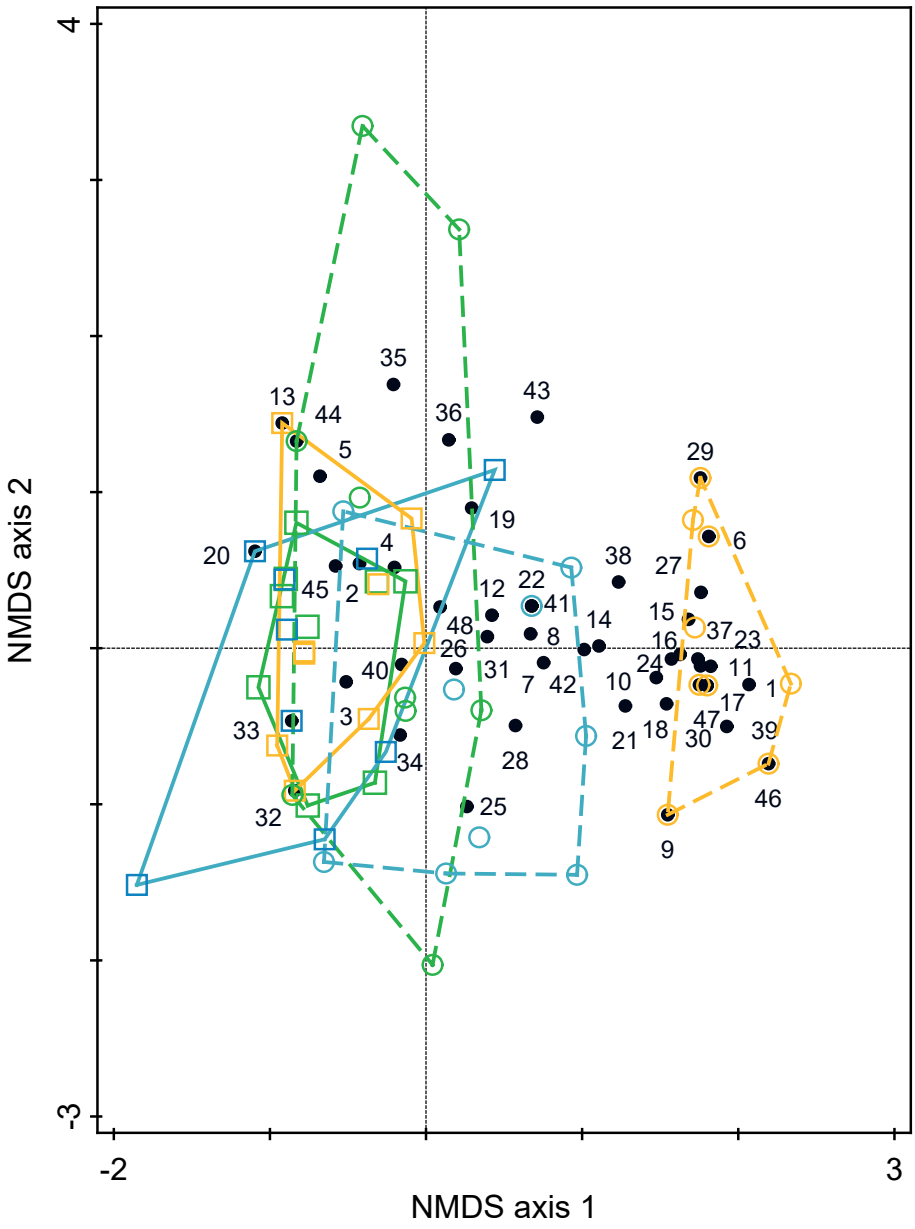

Figure 4.6 Biplot scores for stenotopic early successional beetle species (●) and treatment plots, calculated from Non-Metric Multidimensional Scaling of abundance data at Canvey Wick ground disturbance plots, in 2015 and 2016 (stress value = 0.1448, 3 dimensions). Ordination was based on a matrix of Jaccard dissimilarities, with a maximum of 100 random starts to search for the lowest stress solution. Samples were pooled in each year from three pitfall traps per monitoring plot (n=27), open for sixteen days in May, July, and September. All plots were monitored before management in 2015. Excavator disturbance and scraping treatments were applied in November 2015 (both n=9), and monitoring was repeated in 2016. Management treatments are coded ( ) unmanaged grassland in 2015, ( ) unamanged grassland in 2015, ( ) unmanaged grassland in 2015, ( ) unmanaged grassland in 2016, ( ) excavator disturbance plots in 2016, ( ) scraped plots in 2016. Numerical references for species names: 1 = *Agonum marginatum*, 2 = *Amara aenea*, 3 = *Amara convexior*, 4 = *Amara eurynota*, 5 = *Amara tibialis*, 6 = *Anthicus antherinus*, 7 = *Anthracus consputus*, 8 = *Bembidion lunulatum*, 9 = *Bembidion varium*, 10 = *Bledius gallicus*, 11 = *Bledius opacus*, 12 = *Brachinus crepitans*, 13 = *Calathus ambiguus*, 14 = *Carpelimus corticinus*, 15 = *Carpelimus erichsoni*, 16 = *Carpelimus gracilis*, 17 = *Carpelimus incongruus*, 18 = *Carpelimus rivularis*, 19 = *Curimopsis maritima*, 20 = *Curtonotus convexiusculus*, 21 = *Dyschirius aeneus*, 22 = *Dyschirius luedersi*, 23 = *Dyschirius politus*, 24 = *Elaphrus riparius*, 25 = *Harpalus rubripes*, 26 = *Harpalus tardus*, 27 = *Hippodamia variegata*, 28 = *Leiodes rufipennis*, 29 = *Melanophthalma suturalis*, 30 = *Neobisnius procerulus*, 31 = *Notiophilus substriatus*, 32 = *Notoxus monoceros*, 33 = *Otiorhynchus ovatus*, 34 = *Paederus littoralis*, 35 = *Panagaeus bipustulatus*, 36 = *Paradromius linearis*, 37 = *Philonthus quisquiliarius*, 38 = *Platystethus alutaceus*, 39 = *Platystethus nitens*, 40 = *Poecilus cupreus*, 41 = *Sitona cylindricollis*, 42 = *Sunius propinquus*, 43 = *Syntomus foveatus*, 44 = *Tasgius globulifer*, 45 = *Tasgius winkleri*, 46 = *Thecturota marchii*, 47 = *Thinonoma atra*, 48 = *Xantholinus elegans*.

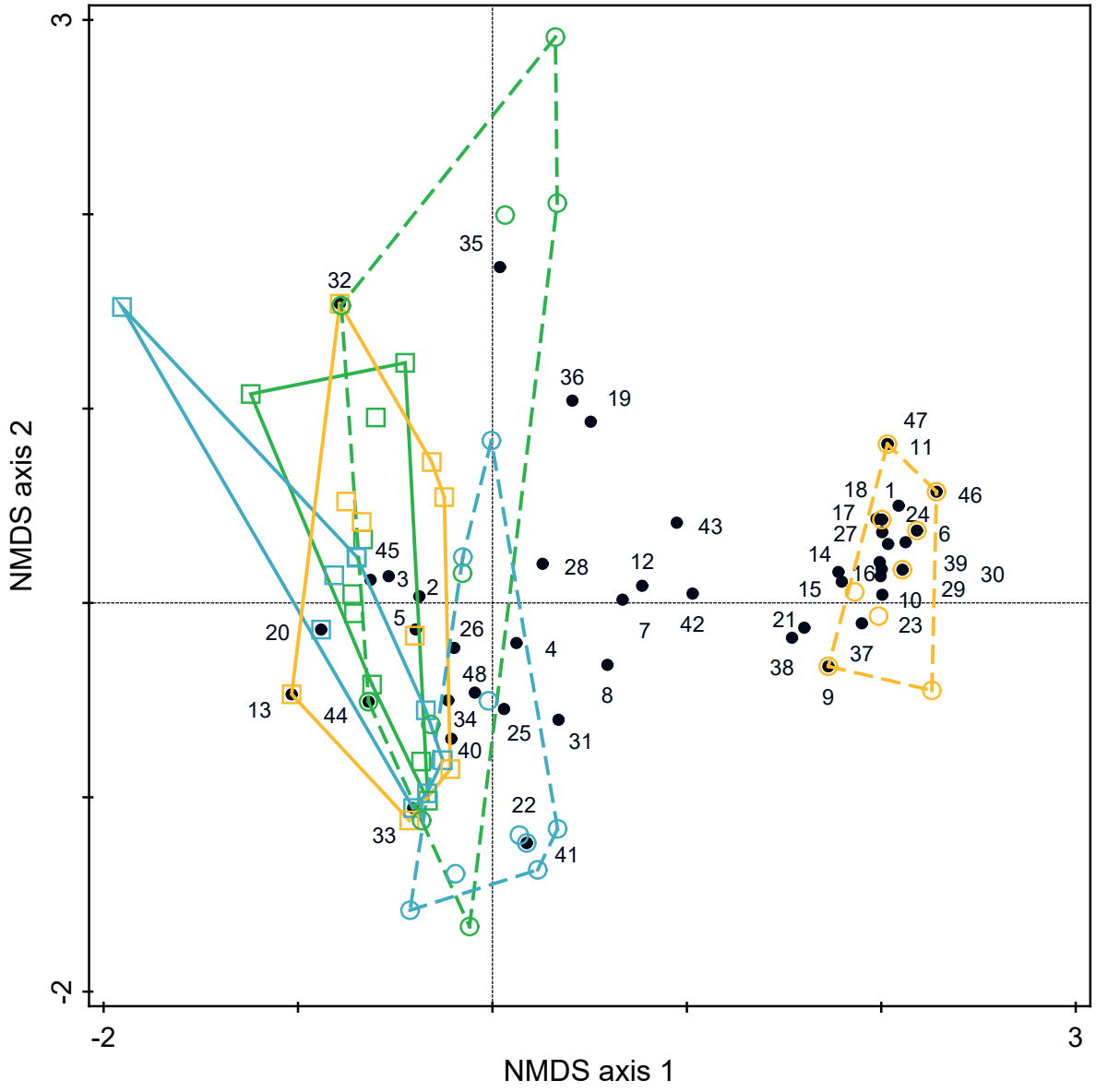

Figure 4.7 Biplot scores for stenotopic early successional beetle species (●) and treatment plots, calculated from Non-Metric Multidimensional Scaling of abundance data at Canvey Wick ground disturbance plots, in 2015 and 2016 (stress value = 0.1295, 3 dimensions). Ordination was based on a matrix of Bray-Curtis dissimilarities, with a maximum of 100 random starts to search for the lowest stress solution. Samples were pooled in each year from three pitfall traps per monitoring plot (n=27), open for 16 days in May, July, and September. All plots were monitored before management in 2015. Excavator disturbance and scraping treatments were applied in November 2015 (both n=9), and monitoring was repeated in 2016. Management treatments are coded ( ) unmanaged grassland in 2015, ( ) unamanged grassland in 2015, ( ) unmanaged grassland in 2015, ( ) unmanaged grassland in 2016, ( ) excavator disturbance plots in 2016, ( ) scraped plots in 2016. Numerical references for species names: 1 = *Agonum marginatum*, 2 = *Amara aenea*, 3 = *Amara convexior*, 4 = *Amara eurynota*, 5 = *Amara tibialis*, 6 = *Anthicus antherinus*, 7 = *Anthracus consputus*, 8 = *Bembidion lunulatum*, 9 = *Bembidion varium*, 10 = *Bledius gallicus*, 11 = *Bledius opacus*, 12 = *Brachinus crepitans*, 13 = *Calathus ambiguus*, 14 = *Carpelimus corticinus*, 15 = *Carpelimus erichsoni*, 16 = *Carpelimus gracilis*, 17 = *Carpelimus incongruus*, 18 = *Carpelimus rivularis*, 19 = *Curimopsis maritima*, 20 = *Curtonotus convexiusculus*, 21 = *Dyschirius aeneus*, 22 = *Dyschirius luedersi*, 23 = *Dyschirius politus*, 24 = *Elaphrus riparius*, 25 = *Harpalus rubripes*, 26 = *Harpalus tardus*, 27 = *Hippodamia variegata*, 28 = *Leiodes rufipennis*, 29 = *Melanophthalma suturalis*, 30 = *Neobisnius procerulus*, 31 = *Notiophilus substriatus*, 32 = *Notoxus monoceros*, 33 = *Otiorhynchus ovatus*, 34 = *Paederus littoralis*, 35 = *Panagaeus bipustulatus*, 36 = *Paradromius linearis*, 37 = *Philonthus quisquiliarius*, 38 = *Platystethus alutaceus*, 39 = *Platystethus nitens*, 40 = *Poecilus cupreus*, 41 = *Sitona cylindricollis*, 42 = *Sunius propinquus*, 43 = *Syntomus foveatus*, 44 = *Tasgius globulifer*, 45 = *Tasgius winkleri*, 46 = *Thecturota marchii*, 47 = *Thinonoma atra*, 48 = *Xantholinus elegans*.

Calodera protensa (269) and *Lamprinodes saginatus* (11) maintained relatively even abundance between treatments. *Helophorus alternans* (17) was particularly associated with scrapes. Two species decreased during the study. *Helops caeruleus* was recorded singly in 2015, though as part of the saproxylic fauna (Fattorini *et al.*, 2015) this was not an indicator of grassland management. *Ocypus fuscatus* decreased from 27 to 14 individuals between 2015 and 2016.

4.3.5.2 Spiders

There was no significant difference in abundance ($\chi^2_{[2, N=27]} = 2.11$, $P = 0.3474$) or species richness ($\chi^2_{[2, N=27]} = 1.97$, $P = 0.6796$) of stenotopic spiders between treatments in 2015 (Table 4.5). Stenotopic spiders differed significantly in abundance ($\chi^2_{[2, N=27]} = 32.81$, $P < 0.0001$) between treatments in 2016, but did not differ in richness ($\chi^2_{[2, N=27]} = 0.63$, $P = 0.7315$). Abundance was significantly higher at control and disturbed plots than scraped plots ($P_{adj} < 0.0001$ in both cases). There was no significant difference in abundance ($\chi^2_{[2, N=27]} = 1.50$, $P = 0.4731$) or species richness ($\chi^2_{[2, N=27]} = 0.17$, $P = 0.917$) of generalist spiders between treatments in 2015. Generalist spiders differed significantly in abundance ($\chi^2_{[2, N=27]} = 348.66.$, $P < 0.0001$) and richness ($\chi^2_{[2, N=27]} = 4.38$, $P = 0.0386$) between treatments in 2016. Abundance was significantly higher at control and disturbed plots than scraped plots ($P_{adj} < 0.0001$ in both cases) and significantly higher at disturbed plots than control plots ($P_{adj} = 0.0007$). Richness was higher at control plots than scraped plots ($P_{adj} = 0.0386$).

NMDS based on presence/absence (Figure 4.8) showed that stenotopic spider communities partially overlapped at treatment and control plots in both years (stress value 0.1184). From 17 stenotopic spider species, only *Clubiona neglecta* (3 specimens) was not recorded in 2016. After the application of treatments, *Argenna subnigra* and *Erigone arctica* were unique to scraped plots, and *Pardosa agrestis* and *Mermessus trilobatus* were unique to disturbed plots, all as single specimens. NMDS based on abundance (Figure 4.9) separated communities at disturbed and scraped plots in 2016, and partially separated control plots (stress value 0.1267). The species with the largest increases at control and disturbed plots were *Drassyllus pusillus* (81 to 170), *Haplodrassus signifer* (11 to 39), *Ozyptila simplex* (61 to 208) and *Xysticus kochi* (3 to 22). Abundance of stenotopic spiders reduced after scraping, although *Arctosa*

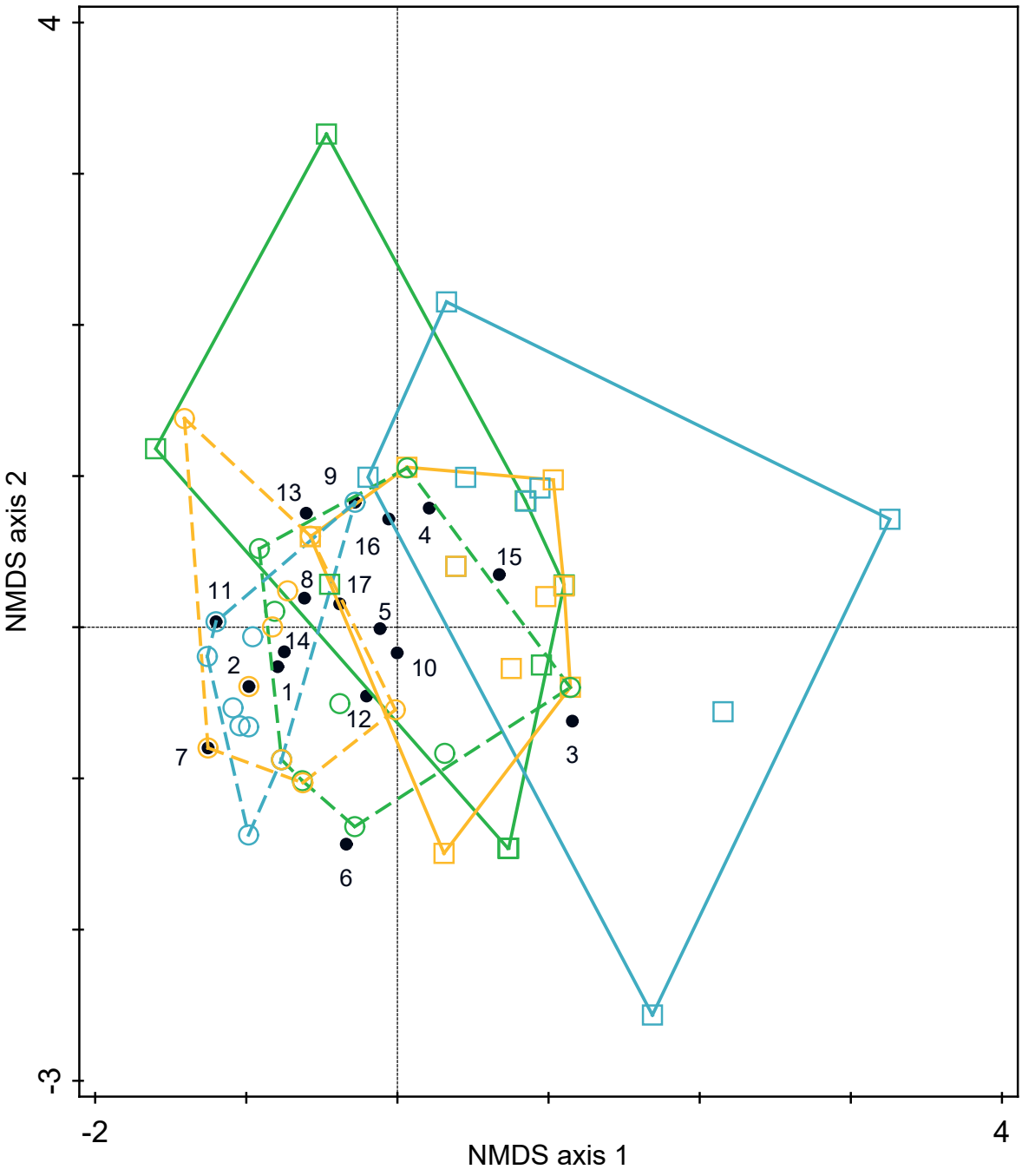

Figure 4.8 Biplot scores for stenotopic early successional spider species (●) and treatment plots, calculated from Non-metric Multidimensional Scaling of presence/absence data at Canvey Wick ground disturbance plots, in 2015 and 2016 (stress value = 0.1184, 3 dimensions). Ordination was based on a matrix of Jaccard dissimilarities, with a maximum of 100 random starts to search for the lowest stress solution. Samples were pooled in each year from three pitfall traps per monitoring plot (n=27), open for 16 days in May, July,and September. All plots were monitored before management in 2015. Excavator disturbance and scraping treatments were applied in November 2015 (both n=9), and monitoring was repeated in 2016. Management treatments are coded ( ) unmanaged grassland in 2015, ( ) unamanged grassland in 2015, ( ) unmanaged grassland in 2015, ( ) unmanaged grassland in 2016, ( ) excavator disturbance plots in 2016, ( ) scraped plots in 2016. Numerical references for species names: 1 = *Arctosa leopardus*, 2 = *Argenna subnigra*, 3 = *Clubiona neglecta*, 4 = *Drassodes cupreus*, 5 = *Drassyllus pusillus*, 6 = *Dysdera crocata*, 7 = *Erigone arctica*, 8 = *Haplodrassus signifer*, 9 = *Mermessus trilobatus*, 10 = *Ozyptila simplex*, 11 = *Pardosa agrestis*, 12 = *Sibianor aurocinctus*, 13 = *Trachyzelotes pedestris*, 14 = *Xysticus kochi*, 15 = *Zelotes apricorum*, 16 = *Zelotes latreillei*, 17 = *Zodarion italicum*.

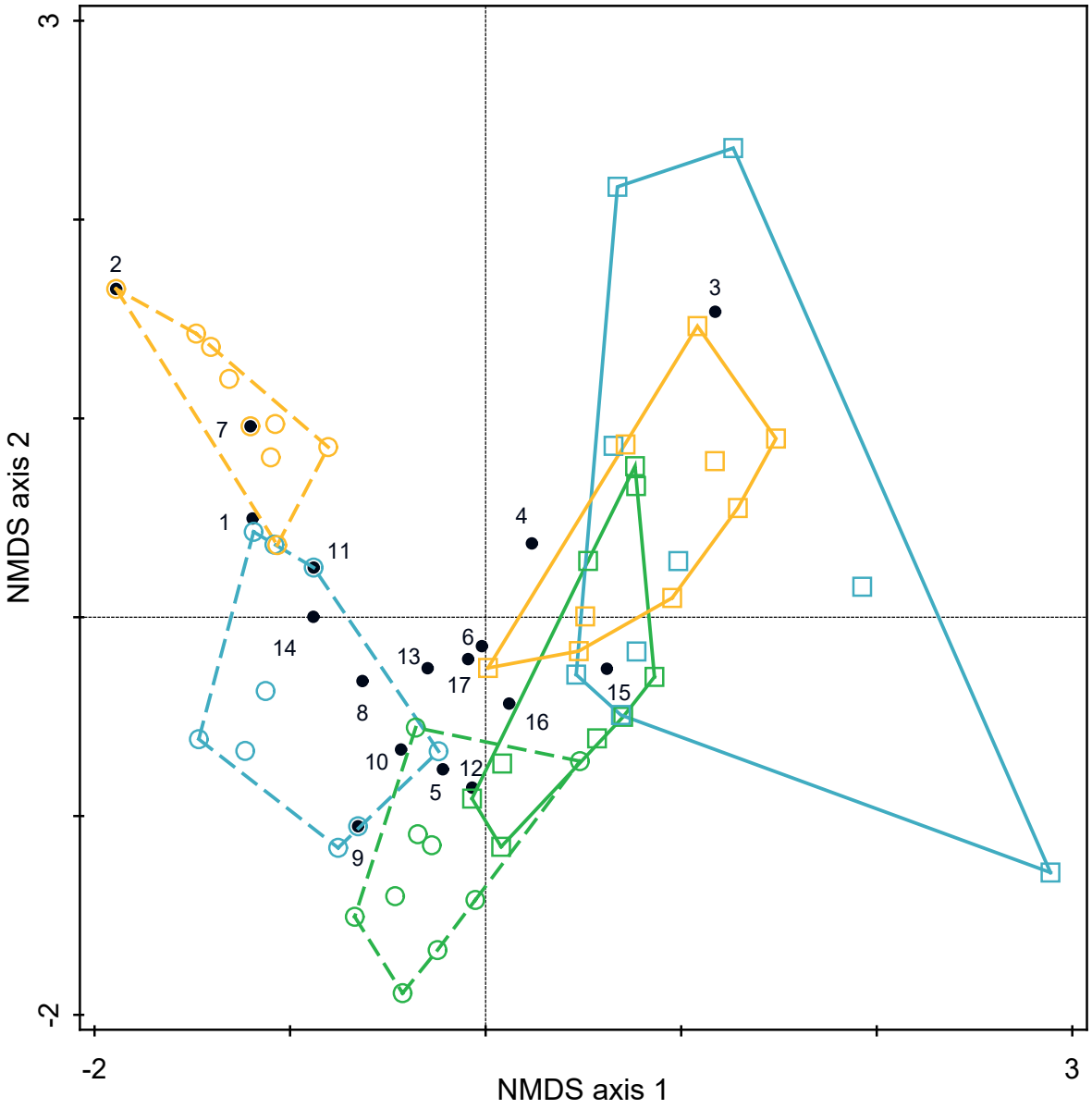

Figure 4.9 Biplot scores for stenotopic early successional spider species (●) and treatment plots, calculated from Non-metric Multidimensional Scaling of presence/absence data at Canvey Wick ground disturbance plots, in 2015 and 2016 (stress value = 0.1267, 3 dimensions). Ordination was based on a matrix of Bray-Curtis dissimilarities, with a maximum of 100 random starts to search for the lowest stress solution. Samples were pooled in each year from three pitfall traps per monitoring plot (n=27), open for 16 days in May, July,and September. All plots were monitored before management in 2015. Excavator disturbance and scraping treatments were applied in November 2015 (both n=9), and monitoring was repeated in 2016. Management treatments are coded ( ) unmanaged grassland in 2015, ( ) unamanged grassland in 2015, ( ) unmanaged grassland in 2015 ( ) unmanaged grassland in 2016, ( ) excavator disturbance plots in 2016, ( ) scraped plots in 2016. Numerical references for species names: 1 = *Arctosa leopardus*, 2 = *Argenna subnigra*, 3 = *Clubiona neglecta*, 4 = *Drassodes cupreus*, 5 = *Drassyllus pusillus*, 6 = *Dysdera crocata*, 7 = *Erigone arctica*, 8 = *Haplodrassus signifer*, 9 = *Mermessus trilobatus*, 10 = *Ozyptila simplex*, 11 = *Pardosa agrestis*, 12 = *Sibianor aurocinctus*, 13 = *Trachyzelotes pedestris*, 14 = *Xysticus kochi*, 15 = *Zelotes apricorum*, 16 = *Zelotes latreillei*, 17 = *Zodarion italicum*.

leopardus (105) was more numerous, compared with disturbed (61) or control plots (10). Two species with conservation status were not classified as part of the disturbance management community. *Centromerus sylvaticus* was recorded as a single individual in 2015 only, and *Agraecina striata* increased almost eight-fold at control plots in 2016.

4.3.6 Response of functional groups

Predatory species did not differ significantly in abundance ($\chi^2_{[2, N=27]} = 0.05$, $P = 1$) or species richness ($\chi^2_{[2, N=27]} = 0.35$, $P = 0.8412$) between treatments in 2015 (Table 4.6). Predatory species differed significantly in abundance ($\chi^2_{[2, N=27]} = 76.70$, $P < 0.0001$) between treatments in 2016, but not in richness ($\chi^2_{[2, N=27]} = 3.74$, $P = 0.154$). Abundance was significantly higher at control and disturbed plots than scraped plots ($P_{adj} < 0.0001$ in both cases), with Staphylinidae (39.4% of predatory taxa) and Lycosidae (26.9%) more than twice as frequent. The three most abundant predatory taxa comprised 29.8% of guild abundance in 2016. The lycosid spider *Pardosa pullata* was between three and five times as abundant at control (328) and disturbed plots (529) compared with scrapes. The staphylinid *Drusilla canaliculata* was more than twice as abundant at control plots (541) than disturbed plots (215), and much less frequent at scrapes (31). The staphylinid *Aleochara brevipennis* was twice as abundant at control plots (99) compared with scrapes, and more than ten times as abundant at disturbed plots (528).

Zoophagous species did not differ significantly in abundance ($\chi^2_{[2, N=27]} = 0.11$, $P = 0.9441$) or species richness ($\chi^2_{[2, N=27]} = 0.35$, $P = 0.8412$) between treatments in 2015. Zoophagous species differed significantly in abundance ($\chi^2_{[2, N=27]} = 9.41$, $P = 0.009$) between treatments in 2016, but not in richness ($\chi^2_{[2, N=27]} = 0.58$, $P = 0.7477$). Abundance was significantly higher at control and disturbed plots than scraped plots ($P_{adj} = 0.0175$ in both cases). The carabids *Acupalpus dubius*, *Agonum emarginatum* and *Pterostichus nigrita* comprised 53% of zoophagous guild abundance in 2016. In combination, these were over three times more abundant at control plots (436) and disturbed plots (406) than scraped plots.

Table 4.6 Mean ± standard error for species richness and abundance of beetle and spider functional groups at Canvey Wick ground disturbance experimental plots in 2015-2016. Generalized linear mixed models (χ^2, F) compared predatory (beetle and spider), zoophagous, phytophagous, and detritivorous groups (beetles only). Treatments were excavator disturbance (n=9), scraping (n=9), and unmanaged grassland controls (n=9). In each year, samples were pooled from three pitfall traps within each monitoring plot, open for 16 days in May, July, and September. Post-hoc tests show Tukey pairwise comparisons after Holm-Šidák adjustment, treatments with shared superscript a-c (ranked highest to lowest) did not differ significantly ($P > 0.05$). Moran's I was calculated to test for spatial correlation in GLMM residuals, based on a matrix of inverse Euclidean distance weights (significance level $P < 0.05$).

Functional group	Year	Measure	Control	Disturbance	Scraping	$GLMM_{[2, N=27]}$	Moran's I_{GLM}
Predatory	2015	Abundance	259.1 ± 28.6 ^{a}	255.8 ± 24.7 ^{a}	265.0 ± 30.5 ^{a}	$\chi^2 = 0.05$	0.8007
		Species richness	41.8 ± 2.1 ^{a}	40.7 ± 2.0 ^{a}	44.4 ± 2.7 ^{a}	$F = 1.60$	0.7429
	2016	Abundance	353.0 ± 34.9 ^{a}	395.2 ± 37.8 ^{a}	170.8 ± 37.8 ^{b}	$\chi^2 = 76.70$	0.7082
		Species richness	49.9 ± 2.3 ^{a}	53.2 ± 1.7 ^{a}	46.8 ± 1.58 ^{a}	$F = 3.74$	0.2006
Zoophagous	2015	Abundance	83.0 ± 17.9 ^{a}	81.1 ± 11.8 ^{a}	89.3 ± 15.0 ^{a}	$\chi^2 = 0.11$	0.2834
		Species richness	13.3 ± 0.9 ^{a}	13.0 ± 0.4 ^{a}	14.0 ± 0.9 ^{a}	$F = 0.35$	0.6069
	2016	Abundance	77.8 ± 11.9 ^{a}	77.0 ± 8.6 ^{a}	48.1 ± 5.3 ^{b}	$\chi^2 = 9.41$	0.2005
		Species richness	13.2 ± 0.7 ^{a}	13.8 ± 0.8 ^{a}	14.6 ± 1.2 ^{a}	$F = 0.58$	0.9939
Phytophagous	2015	Abundance	135.0 ± 22.6 ^{a}	127.1 ± 22.7 ^{a}	139.7 ± 17.1 ^{a}	$\chi^2 = 0.22$	0.4160
		Species richness	9.1 ± 0.7 ^{a}	9.3 ± 0.8 ^{a}	9.4 ± 0.6 ^{a}	$F = 0.06$	0.9092
	2016	Abundance	89.4 ± 15.0 ^{a}	48.8 ± 6.1 ^{b}	52.4 ± 5.3 ^{c}	$\chi^2 = 13.86$	0.5066
		Species richness	9.9 ± 0.8 ^{a}	12.6 ± 1.0 ^{a}	12.4 ± 0.9 ^{a}	$F = 3.62$	0.1262
Detritivorous	2015	Abundance	10.1 ± 2.6 ^{a}	9.6 ± 2.2 ^{a}	7.7 ± 1.8 ^{a}	$F = 1.05$	0.4055
		Species richness	4.3 ± 0.9 ^{a}	4.2 ± 0.5 ^{a}	3.1 ± 0.3 ^{a}	$\chi^2 = 2.19$	0.1667
	2016	Abundance	8.1 ± 1.4 ^{a}	10.9 ± 1.7 ^{a}	9.8 ± 1.6 ^{a}	$F = 3.72$	0.3658
		Species richness	5.1 ± 0.7 ^{a}	4.9 ± 0.4 ^{a}	5.4 ± 0.7 ^{a}	$F = 0.27$	0.7469

Detritivorous species did not differ significantly in abundance ($\chi^2_{[2, N=27]}$ = 1.05, *P* = 0.5916) or species richness in 2015 ($\chi^2_{[2, N=27]}$ = 2.19, *P* = 0.3338), and did not differ significantly in abundance ($\chi^2_{[2, N=27]}$ = 1.05, *P* = 0.5916) or richness in 2016 ($\chi^2_{[2, N=27]}$ = 3.72, *P* = 0.156). Three beetles accounted for 60.2% of detritivore abundance in 2016. *Megasternum concinnum* was six times as abundant at control plots (18) compared with scraped plots (3), and 14 times as abundant at disturbed plots (42). *Anotylus rugosus* was more frequent at disturbed (17) and scraped plots (20) than control plots (11), although the difference was relatively small. *Aphodius plagiatus* was absent from control plots, and occurred in similar numbers at disturbed (11) and scraped plots (14).

Phytophagous species did not differ in abundance ($\chi^2_{[2, N=27]}$ = 0.21, *P* = 0.8973) or richness ($\chi^2_{[2, N=27]}$ = 0.06, *P* = 0.9724) between treatments in 2015. Phytophagous species differed significantly in abundance ($\chi^2_{[2, N=27]}$ = 13.86, *P* = 0.001) between treatments in 2016, but did not differ in richness ($\chi^2_{[2, N=27]}$ = 3.62, *P* = 0.1636). Abundance was significantly higher at controls than disturbed (P_{adj} = 0.0021) or scraped plots (P_{adj} = 0.0056). Three taxa accounted for 70% of phytophagous trophic group abundance in 2016. *Agriotes lineatus* was five to six times more abundant at control (178) and disturbed plots (220) than scraped plots. *Amara communis* and *Amara lunicollis* in combination were five times more abundant at disturbed plots (81) and 35 times more abundant at control plots (536).

4.3.7 Indicator species

There were no separate indicator species for control or disturbed plots, although in combination these treatments produced nine indicators (Table 4.7). These were two staphylinids, three carabids, one silphid, and one spider. There were three indicator species for disturbed and scraped plots in combination. These were one chrysomelid and two spiders. Scraped plots also produced seven indicator species. These were four staphylinids, two carabids, and one spider.

4.3.8 Emergent group results

4.3.8.1 Carabid emergent groups

Cluster analysis revealed six groups for carabid species at a cut-off level S = 0.54 (Figure 4.10). Carabids in emergent group C1 (n = 8) were macropterous,

Table 4.7 List of significant indicator species (P < 0.0001, Monte Carlo test, 4,999 permutations) for carabids, staphylinids, and spiders at Canvey Wick ground disturbance plots in 2016, based on occurrence of more than 10 individuals in treatments separately, or in combination. Fidelity describes limitation in species occurrence among treatments, or treatment groups (1=single). Constancy describes frequency of occurrence within a treatment, or treatment groups (1=constant). Plus symbol (+) by species name denotes status as a stenotopic indicator of early successional habitat in this study.

Invertebrate group	Treatment(s)	Taxon	Individuals	Fidelity	Constancy	Index value
Coleoptera	Control + disturbance	*Amara lunicollis*	378	0.9818	1.0000	0.991
		Amara communis	239	0.9676	1.0000	0.984
		Silpha tristis	418	0.9457	1.0000	0.972
		Aleochara brevipennis	627	0.9372	1.0000	0.968
		Staphylinus dimidiaticornis	164	0.9880	0.9444	0.966
		Pterostichus niger	33	1.0000	0.7778	0.882
	Disturbance + scraping	*Chaetocnema hortensis*	78	0.9873	1.0000	0.994
	Scraping	*Bledius gallicus*+	143	0.9862	1.0000	0.993
		Elaphrus riparius+	70	0.9859	1.0000	0.993
		Carpelimus gracilis+	79	0.9753	1.0000	0.988
		Bembidion quadrimaculatum	74	0.9487	1.0000	0.974
		Carpelimus corticinus+	83	0.9121	1.0000	0.955
		Neobisnius procerulus+	45	0.9375	0.8889	0.913
Araneae	Control + disturbance	*Alopecosa pulverulenta*	439	0.9502	1.0000	0.975
		Ozyptila simplex	196	0.9423	1.0000	0.971
		Agraecina striata	107	0.9817	0.9444	0.963
	Disturbance + scraping	*Erigone atra*	169	0.9494	1.0000	0.974
		Arctosa leopardus+	166	0.9432	1.0000	0.971
	Scraping	*Erigone dentipalpis*	138	0.8846	1.0000	0.941

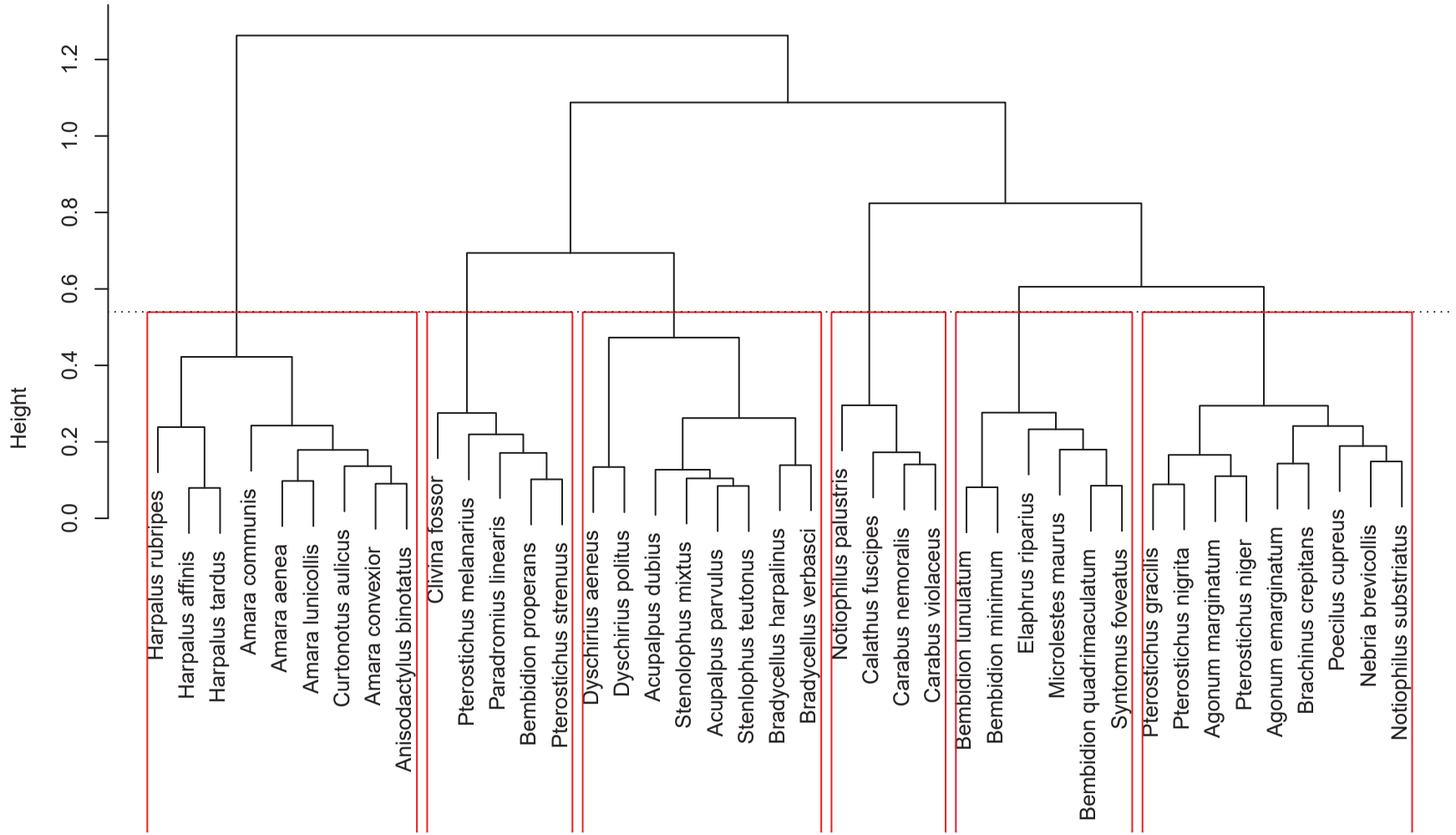

Figure 4.10 Cluster dendrogram for carabid species (n=40) from experimental ground disturbance plots at Canvey Wick in 2016. This was based on morphological traits for single male and female specimens, with the dissimilarity matrix calculated from Gower's coefficient (1971) for quantitative and qualitative measures. Hierarchical clustering was completed by agglomeration, with Ward's criterion (1963) to determine the decrease in variance for the cluster being merged. The number of emergent groups (n=6) was determined subjectively by visual examination of the clustering solution (S = 0.54).

with longer hind tibiae and shorter hind tarsi. These did not differ significantly in abundance between treatments ($\chi^2_{[2, N=27]}$ = 2.13, P = 0.3455). Carabids in emergent group C2 (n = 8) were mostly larger species, with longer antennae and shorter hind femora relative to length of leg. These differed significantly in abundance between treatments ($\chi^2_{[2, N=27]}$ = 25.55, P < 0.0001). Abundance was significantly higher at control and disturbed plots than scraped plots (P_{adj} < 0.0001 in both cases). Carabids in emergent group C3 (n = 9) had smaller eyes relative to head width, longer trochanters relative to the hind femur, and wider pronotums. These differed significantly in abundance between treatments ($\chi^2_{[2, N=27]}$ = 146.70, P < 0.0001). Abundance was significantly higher at control and disturbed plots than scraped plots, and greater at control plots than disturbed plots (P_{adj} < 0.0001 in all cases).

Carabids in emergent group C4 (n = 7) were smaller species, with longer antennae and longer femora relative to length of leg. These differed significantly in abundance between treatments ($\chi^2_{[2, N=27]}$ = 239.86, P < 0.0001) (Table 4.8). Abundance was significantly higher at scraped plots than controls or disturbed plots (P_{adj} < 0.0001 in both cases). Carabids in emergent group C5 (n = 6) were dimorphic species with shorter hind tibiae and longer hind tarsi. These differed significantly in abundance between treatments ($\chi^2_{[2, N=27]}$ = 64.52, P < 0.0001). Abundance was significantly higher at control and disturbed plots than scraped plots (P_{adj} < 0.0001 in both cases) and significantly higher at control plots than disturbed plots (P_{adj} = 0.0158). Carabids in emergent group C6 (n = 4) were brachypterous, mostly with larger body size, longer antennae and longer hind femora. These differed significantly in abundance between treatments ($\chi^2_{[2, N=27]}$ = 6.26, P = 0.0437). Abundance was significantly higher at disturbed plots compared with scraped plots (P_{adj} = 0.0439).

4.3.8.2 Staphylinid emergent groups

Cluster analysis revealed seven groups for the staphylinid species at a cut-off level S = 0.38 (Figure 4.11). Staphylinids in emergent group S1 (n = 12) were smaller species with relatively long antennae. These differed significantly in abundance between treatments ($\chi^2_{[2, N=27]}$ = 40.50, P < 0.0001) (Table 4.8). Abundance was significantly higher at disturbed plots than control or scraped plots (P_{adj} < 0.0001 in both cases). Staphylinids in emergent group S2 (n = 7)

Table 4.8 Mean ± standard error for abundance of carabid and staphylinid emergent groups at Canvey Wick ground disturbance experimental plots in 2016. Generalized linear mixed models (χ^2, F) compare emergent groups at grassland controls (n=9), excavator disturbance (n=9), and scraping (n=9) treatments. In each year, samples were pooled from three pitfall traps within each monitoring plot, open for 16 days in May, July, and September. Emergent groups were defined from hierarchical clustering of morphological characteristics. Post-hoc tests show Tukey pairwise comparisons after Holm-Šidák adjustment, treatments with shared superscript a-c (ranked highest to lowest) did not differ significantly ($P > 0.05$).

Coleoptera family	Emergent group	Control	Disturbance	Scraping	$GLMM_{[2, N=27]}$	Moran's I_{GLM}
Carabidae	C1	11.3 ± 2.0 [a]	12.1 ± 2.3 [a]	8.2 ± 2.0 [a]	$\chi^2 = 2.13$	0.2316
	C2	45.9 ± 9.8 [a]	46.7 ± 6.7 [a]	15.0 ± 2.6 [b]	$\chi^2 = 25.55$	0.7613
	C3	62.7 ± 13.2 [a]	11.1 ± 1.5 [b]	3.8 ± 1.0 [c]	$\chi^2 = 146.70$	0.8759
	C4	0.9 ± 0.3 [b]	1.4 ± 0.4 [b]	17.8 ± 2.4 [a]	$F = 239.86$	0.5299
	C5	15.9 ± 2.9 [a]	11.7 ± 1.2 [b]	4.3 ± 0.7 [c]	$F = 64.52$	0.2025
	C6	3.8 ± 0.8 [ab]	4.9 ± 1.3 [a]	1.9 ± 0.6 [b]	$\chi^2 = 6.26$	0.6778
Staphylinidae	S1	35.7 ± 7.4 [b]	81.2 ± 17.0 [a]	28.9 ± 5.3 [b]	$\chi^2 = 40.50$	0.9881
	S2	1.3 ± 0.6 [c]	3.0 ± 0.8 [b]	30.2 ± 3.4 [a]	$\chi^2 = 400.35$	0.9741
	S3	60.3 ± 3.3 [a]	23.9 ± 6.8 [b]	3.9 ± 0.7 [c]	$\chi^2 = 72.35$	0.2810
	S4	16.1 ± 2.7 [b]	25.0 ± 3.2 [a]	17.7 ± 3.3 [ab]	$\chi^2 = 8.46$	0.0756
	S5	0.3 ± 0.2 [a]	0.5 ± 0.3 [a]	1.1 ± 0.3 [a]	$F = 4.40$	0.1491
	S6	18.4 ± 3.1 [a]	14.6 ± 2.4 [a]	1.7 ± 0.6 [b]	$\chi^2 = 69.91$	0.1201
	S7	6.6 ± 1.2 [a]	5.7 ± 0.8 [a]	1.4 ± 0.4 [b]	$F = 35.35$	0.5622

Figure 4.11 Cluster dendrogram for staphylinid species (n=44) from experimental ground disturbance plots at Canvey Wick in 2016. This was based on morphological traits for single male and female specimens, with the dissimilarity matrix calculated from Gower's coefficient (1971) for quantitative and qualitative measures. Hierarchical clustering was completed by agglomeration, with Ward's criterion (1963) to determine the decrease in variance for the cluster being merged. The number of emergent groups (n=7) was determined subjectively by visual examination of the clustering solution (S = 0.38).

were smaller species with short fore and hind tarsi. These differed significantly in abundance between treatments ($\chi^2_{[2, N=27]} = 400.35$, $P < 0.0001$). Abundance was significantly higher at scraped plots than control or disturbed plots ($P_{adj} < 0.0001$ in both cases) and significantly higher at disturbed plots than control plots ($P_{adj} = 0.019$). Staphylinids in emergent group S3 (n = 3) had longer pronotums and brachypterous or dimorphic wing morphology. These differed significantly in abundance between treatments ($\chi^2_{[2, N=27]} = 72.35$, $P < 0.0001$). Abundance was significantly higher at control and disturbed plots than scraped plots ($P_{adj} < 0.0001$ in both cases) and significantly higher at control plots than disturbed plots ($P_{adj} = 0.0004$).

Staphylinids in emergent group S4 (n = 7) had longer pronotums, longer elytra, and flatter eyes. These differed significantly in abundance between treatments ($\chi^2_{[2, N=27]} = 8.46$, $P < 0.0145$). Abundance was significantly higher at disturbed plots than control plots ($P_{adj} = 0.0197$). Abundance was also significantly higher at disturbed plots than scraped plots but the difference was not significant ($P_{adj} = 0.0655$). Staphylinids in emergent group S5 (n = 3) had short mandibles, long hind tarsi and long hind trochanters. These did not differ significantly in abundance between treatments ($\chi^2_{[2, N=27]} = 4.40$, $P = 0.1111$). Staphylinids in emergent group S6 (n = 11) were larger species with short front tibiae. These differed significantly in abundance between treatments ($\chi^2_{[2, N=27]} = 69.61$, $P < 0.0001$). Abundance was significantly higher at control and disturbed plots than scraped plots ($P_{adj} < 0.0001$ in both cases). Staphylinids in emergent group S7 (n = 4) had larger eyes, shorter trochanters, and relatively short antennae. These differed significantly in abundance between treatments ($\chi^2_{[2, N=27]} = 35.35$, $P < 0.0001$). Abundance was significantly higher at control and disturbed plots than scraped plots ($P_{adj} < 0.0001$ in both cases).

4.4 Discussion

4.4.1 Habitat characteristics

This study focused on the effects of two ground disturbance techniques in an area dominated by perennial graminoids, that was previously used for disposal of sandy river dredgings. Scraped plots had more than 90% bare ground after seven months, compared with less than 1% at plots disturbed by an excavator. The mean vegetation height was less than 2 cm at scrapes, compared with over

15 cm at disturbed plots. At sandy grassland within conifer plantations, there was 30% bare ground in the year after discing, rising to 60% for more disruptive ploughing (Pedley *et al.*, 2013). The rapid revegetation after excavator disturbance in the present study was possibly linked to hydrology. There was surface groundwater for five months during winter 2015/2016, up to 9 cm deep at scrapes and at least saturating the soil at disturbed and control plots. Moisture levels at the disturbed plots probably encouraged regeneration from plant rhizomes and the soil seedbank. Seedling recruitment at dunes coincided with periods of high moisture availability (Maun, 1994). Standing water at scrapes may have inhibited germination, as species richness and biomass of terrestrial plants was reduced after longer duration flooding in experimental wetland microcosms (Casanova and Brock, 2000).

4.4.2 Stenotopic species

Scraping was immediately beneficial for stenotopic beetles, in contrast to a study of carabids and spiders where a specialist carabid assemblage developed after two years (Pedley *et al.*, 2013). This emphasises the need to assess management outcomes for multiple taxonomic groups, as there were species in eight families for the early successional management community in the present study, including 40% staphylinids. Excavator disturbance did not produce a significant increase in stenotopic early successional beetles. There is mixed evidence of the effectiveness of less intensive ground disturbance techniques. At a Dutch heathland, psammophilous carabids were recorded two years after sod-cutting to expose mineral soil (Schirmel, 2010). Less intensive ‘choppering’ left the O-horizon undisturbed, and most colonising carabids were eurytopic species (Schirmel, 2010). Shallow excavator disturbance (< 30 cm) in sandy calcareous grassland produced a distinct beetle assemblage, although species with conservation status were associated with deep disturbance (< 1 m) (Ödman *et al.*, 2011). Abundance of specialist early successional carabids at a forestry trackway network was enhanced within a year by discing and ploughing, although species richness only increased two years after ploughing and turf stripping (Pedley *et al.*, 2013). In the present research, only one year was available for monitoring after the treatments were applied, so it is possible that species richness or abundance might increase at the excavator disturbance plots in future.

There were limited differences in species composition for stenotopic spiders. It is likely that some species occurred in 2015 as casual wanderers from more suitable habitat, or persisted at low population density in the grassland before disturbance. Stenotopic spiders were least abundant on scraped plots, which is consistent with previous research (Pedley *et al.*, 2013). There is evidence that some specialist spiders have more limited capacity to colonise disturbed habitats (Bonte *et al.*, 2003b). An interesting feature of the present study was that abundance of stenotopic spiders more than doubled at control and disturbed plots in 2016, compared with 2015. For species with conservation status in the same period, abundance of spiders increased at control and disturbed plots, and abundance of beetles increased at all treatments. It is normal to attempt to eliminate the effect of adjacent treatments with experimental design, although that was not possible in the area available for this research. Some species may have benefited from the creation of a mosaic of habitat patches, but this cannot be separated from inter-annual variations in population. The patterns of occurrence for priority species also demonstrated that disturbance in areas dominated by perennial graminoids at Canvey Wick did not reduce existing conservation interest for beetles or spiders.

4.4.3 Indicator species

Nine indicator species were associated with control and disturbed plots. *Pterostichus niger* is a large carabid typical of undisturbed grasslands (Rushton *et al.*, 1989; Blake *et al.*, 1994). Although the disturbed plots had no vegetation immediately after treatment, by the following July grass cover was nearly complete. Given the small plots, it is unsurprising that species characteristic of undisturbed habitats were also active on adjacent disturbed plots once vegetation re-established. The carabids *Amara lunicollis* and *Amara communis* are less strongly associated with ruderal weeds than other Zabrini, and were reported as characteristic of mesic floodplain grasslands (Gerisch *et al.*, 2006). The silphid *Silpha tristis* was 55 times more abundant in undisturbed reedbeds in the Camargue, southern France, compared with reedbeds cut in winter (Schmidt *et al.*, 2005). In wing-dimorphic populations, adults and larvae were predaceous on invertebrates in soil and litter (Ikeda *et al.*, 2007) so the scraping treatment was probably unfavourable.

Aleochara brevipennis is part of a staphylinid genus that parasitises Diptera, particularly *Delia* (Anthomyiidae) (Hemachandra *et al.*, 2007). The life cycle of *Delia platura* is typical, with larvae developing in plant seeds or emerging cotyledons, especially among decaying organic matter (Gregory and Musick, 1976). *Staphylinus dimidaticornis* was an indicator of meadows and old pastures, and did not occur in silage fields or cereal crops (Good and Giller 1991). Three spiders were also indicators. The lycosid *Alopecosa pulverulenta* has a wide ecological amplitude in open habitats (Kronestedt, 1990) and was an abundant species at an inland dune complex (Buchholz, 2009). *Ozyptila simplex* is a generally sluggish and retiring ambush hunter found amongst low vegetation (Roberts, 1985). Abundance of this species was correlated with increasing grass cover at calcareous dune grasslands (Bonte *et al.*, 2000). The liocranid *Agraecina striata* is hygrophilous, with no clear preference among vegetated habitats (Harvey *et al.*, 2002). It has been recorded as an indicator of lowland raised bogs (Scott *et al.*, 2006) and calcareous fens (Štokmane *et al.*, 2013).

There were three indicator species associated with disturbed and scraped plots. The chrysomelid beetle *Chaetocnema hortensis* is polyphagous, with larvae feeding on grasses (Poaceae) (Hubble, 2012). There is no clear ecological link to plots with reduced density of vegetation and litter, although this might relate to microhabitat preference. The lycosid spider *Arctosa leopardus* has been recorded in early stages of succession at an artificial lake island (Komposch, 1996), on urban green roofs (Kadas, 2006), and in restored wet heathland (Cristofoli *et al.*, 2010). The linyphiid spider *Erigone atra* is a eurytopic species typical of disturbed habitats (Downie *et al.*, 2000). It is often found in association with *Erigone dentipalpis*, although some studies have found small differences in microhabitat utilisation. Webs of *Erigone atra* were usually higher in pasture vegetation compared with *Erigone dentipalpis*, which was almost only found in short-grazed areas during spring and summer (de Keer and Maelfait, 1988). This suggests that greater revegetation following the disturbance treatment would also be colonised by *Erigone atra*, although this species was more abundant at scraped plots overall.

Seven indicator species were associated with scraped plots, including the geophilic staphylinids *Bledius gallicus*, *Carpelimus gracilis* and *Carpelimus*

corticinus. Adult *Bledius* have the prothorax attached to the hind body on a pedicel, allowing considerable flexibility (Lott, 2009). Burrows are excavated with their mandibles, and spine-like setae along the protibiae provide support when digging (Herman, 1986). Adult *Carpelimus* burrow in soft silt and sand and have a similar stocky body (De Marzo, 2016). The staphylinid *Neobisnius procerulus* is mainly associated with exposed sediments in secondary habitats in the UK (Lott, 2009) but in Europe also occurs along river floodplains (Derunkov, 2004). Two carabid indicators were *Elaphrus riparius* and *Bembidion quadrimaculatum*. *Elaphrus* species are well-adapted to hunting in open areas, with long tarsomeres and metafemora, small metatrochanters, and large surface area of the eyes (Forsythe, 1981; Bauer *et al.*, 1998). *Bembidion quadrimaculatum* is typical of early successional communities, with broad preferences including urban brownfields (Small *et al.*, 2003) and agricultural land (Melnychuk *et al.*, 2003). The only spider indicator was the linyphiid *Erigone dentipalpis*. This species was more dominant at arable farms in sparsely vegetated spring-sown crops, compared to autumn-sown crops with higher levels of plant biomass (Downie *et al.*, 2000).

4.4.4 Functional groups

There were no significant differences in species richness or abundance of functional groups in 2015, before the treatments were applied. There were also no significant differences in species richness or abundance of detritivores in 2016, although there were some patterns within this guild. Coprophagous species were most frequent at disturbed plots and scrapes, including *Megasternum concinnum*, *Aphodius plagiatus* and *Aphodius sphacelatus*. This was probably linked to rabbit activity, as droppings were frequent at scrapes and disturbed plots, and rare at controls. *Megasternum concinnum* was also more abundant in cattle-grazed pastures with increased levels of dung, compared with fenced field margins (Anderson *et al.*, 2013). Helophoridae were more frequent on scrapes. Adults in this genus feed on decaying vegetable matter, but larvae of most species are predatory at water margins (Foster *et al.*, 2014). The scrapes retained surface groundwater until late April 2016, in greater volume and for longer than the control or disturbed plots. It is likely that the adults were recorded in association with ephemeral breeding sites (Florencio *et al.*, 2009).

Reduced abundance of phytophagous species at scrapes was predictable, as species richness of forbs and graminoids was lower (Woodcock *et al.*, 2009). *Agriotes lineatus* was classified as phytophagous as adults feed on pollen, although abundance at disturbed and control plots in this study was probably associated with breeding activity, as larvae can be abundant at the roots of grasses (Benefer *et al.*, 2010). A similar pattern for *Amara lunicollis* and *Amara convexior* reflects the importance of plant seeds for larval development and adult diet in this genus (Fawki and Toft, 2005), and scraped plots lacked these resources. For other phytophagous families, including Apionidae, Curculionidae, and Chrysomelidae, abundance was greatest at disturbed plots and positively correlated with plant species richness. Conservation field margins with higher species richness of grasses and forbs also supported greater numbers of insect herbivores (Anderson *et al.*, 2013). There was a distinct group of phytophagous staphylinids predominately associated with scraped plots. The diet for species of *Bledius* and *Carpelimus* is algae (Herman, 1986; Catenazzi and Donnelly, 2007) and adults dig nest burrows, particularly in areas of damp sand and silt.

Predatory and zoophagous species were more abundant at control and disturbed plots. Large and medium-sized predatory and zoophagous beetles were also abundant in tall, ungrazed saltmarsh grasslands (Ford *et al.*, 2013). Of abundant zoophagous taxa, *Agonum emarginatum* was a dominant species in floodplain grasslands (Gerisch, 2014) and *Pterostichus nigrita* was ubiquitous in upland grasslands (Rushton *et al.*, 1989). Amelioration of temperature, humidity, and cover from enemies creates a favourable environment for many arthropod predators in tall swards (Morris, 2000). Prey availability was probably reduced at scrapes, and grasslands are frequently associated with abundant Collembola (García-Tejero and Taboada, 2016) and Acari (Curry, 1969). The pattern was different for Linyphiidae, which were around twice as abundant at scraped plots (415) than control (164) or disturbed plots (237). Species included *Erigone dentipalpis* and *Oedothorax apicatus*, which are frequent aeronaut colonists of disturbed habitats (Schmidt and Tscharntke, 2005). These were also part of the agriobiont spider community characteristic of arable fields (Samu and Szinetár, 2002).

4.4.5 Emergent group morphology

4.4.5.1 Carabid emergent groups

Emergent group C3 was partly defined by a long metatrochanter relative to the metafemur. This was also a feature of 'Group II' wedge-pushing carabids (Forsythe, 1981). Species with larger metatrochanters produce greater downwards force on the metatarsus and upwards force on the elytra, and are more able to press forwards in confined spaces (Evans, 1977). The more limited grouping of *Harpalus* and *Amara* in the present study was based on the shared characteristics of wide pronota and smaller eyes. These were linked in a morphological study of Scottish carabids, in addition to the length of the metatrochanter and width of the metafemur (Ribera *et al.*, 1999). This was interpreted as a probable phylogenetic effect, although potentially developed independently as the genera do not form a monophyletic group (Ribera *et al.*, 1999). The association of emergent group C3 with more vegetated plots fits with the morphological adaptation of the metatrochanter, although the species are also partly graminivorous (Talarico *et al.*, 2016) and scraped plots mostly lacked seed resources.

Leg morphology was important in defining two other emergent groups, in combination with body size. Carabids in emergent group C4 were mostly smaller with relatively longer metafemora, compared to larger species in emergent group C2 with shorter metafemora. Group C2 was significantly more abundant at control and disturbed plots, whereas C4 was the only emergent group particularly associated with scraped plots. Coleoptera with longer legs were associated with weaker pushing but faster running (Evans and Forsythe, 1984) which is likely to be an advantage for prey capture in open habitats. Carabids feeding on Collembola had longer femora than species feeding on plant materials, in addition to longer tarsi (Ribera *et al.*, 1999). The association of smaller species with newly scraped plots is consistent with previous research. Smaller carabids were more frequent in ploughed agricultural fields (Ribera *et al.*, 2001) and were also associated with greater management intensity in grasslands (Blake *et al.*, 1994). Insectivorous grassland birds preferred intermediate-sized invertebrate prey (Kaspari and Joern, 1993) and larger species have more chance of escaping predators in dense vegetation (Lawton, 1983).

Two emergent groups were defined by wing morphology. Dimorphic species in group C5 had varied morphological characteristics and were particularly associated with control and disturbed plots. Macropterous individuals of dimorphic species might be well-represented after disturbance, but subsequent development of stable habitat should favour brachypterous specimens. The proportion of brachypterous individuals in a newly established population of *Pterostichus melanarius* increased from 39% in 1991 to 57% in 1997 (Niemelä and Spence, 1999). Measuring wing length or area for captured specimens might enable more accurate assessment of dispersal traits, rather than the classification from literature in this study. Species in emergent group C6 were brachypterous and except for *Notiophilus palustris*, shared larger body size. This supports the theory of k-selection in stable communities, where species develop traits that improve competitive ability such as larger body size, rather than wing development for dispersal (Southwood, 1988). The inclusion of *Notiophilus palustris* is a result of the hierarchical clustering method. Groups are established by maximising a local criterion, which may separate broader morphological groups in early divisions (Boesch, 1977).

4.4.5.2 Staphylinid emergent groups

It is difficult to place the staphylinid emergent groups in context, as there are few ecomorphological studies for this family. Staphylinids in emergent group S5 were in the tribes Tachyporini and Mycetoporini (subfamily Tachyporinae). These shared short mandibles, long hind tarsi and long hind trochanters. There is evidence from gut dissections that *Tachyporus* adults fed on fungus in cereal crops, as well as aphids (Sunderland, 1975). Based on adult activity, *Sepedophilus* were primarily mycophagous, exploiting fungal mycelia (Hammond, 1972). It is likely that the smaller mandibles are a morphological adaptation for this diet. The long trochanters in this group probably modify locomotion in a similar way to carabids, aiding movement in confined spaces (Forsythe, 1981). It is surprising that this group was most frequent on scraped plots, although the number of specimens was low and this may not truly reflect habitat utilisation.

Larger species in emergent group S6 were associated with vegetated control and disturbed plots, as observed for carabids. Larger staphylinids were also more frequent at low levels of flood disturbance (Fournier *et al.*, 2015) and were

associated with scattered shrubs rather than grassland in wood pasture (García-Tejero and Taboada, 2016). Staphylinids in group S6 also shared relatively shorter protibiae but this is difficult to interpret in terms of functional significance. An extra-coxal projection in *Philonthus marginatus* was interpreted functionally for predatory striking (Betz and Mumm, 2001). It was also found in a preliminary survey of 14 species of Staphylininae and Paederinae, and could be an ancestral adaptation for prey capture or handling (Betz and Mumm, 2001). The shortened tibiae may be part of a wider modification of the forelegs, as the fore coxae of *Philonthus marginatus* were significantly elongated, voluminous and coniform, although many Staphylininae capture prey directly in their mandibles (Betz and Mumm, 2001).

Staphylinids in emergent group S7 were separated by larger eyes, shorter trochanters and relatively short antennae. These were in the subfamilies Steninae and Paederinae and were most frequent at disturbed plots. The genus *Stenus* is characterised by large and prominent eyes (Lott and Anderson, 2011). These assist with location of prey that is attacked by extension of a specialised adhesive buccal apparatus, or captured directly in the mandibles (Betz, 1996). Within this pattern, *Stenus comma* preferred areas of bare ground and had widely protruding eyes with greater density of ommatidia, in addition to long legs and slender tarsi (Betz, 1998). *Stenus pubescens* was less active and had flatter eyes with fewer ommatidia, occurring among wet plant debris (Betz, 1998). In this study, *Stenus pusillus* and *Stenus brunnipes* have shorter hind tarsi and would be expected amongst vegetation. *Stenus clavicornis* has longer tarsi but not to the same degree as *Stenus comma*. *Paederus littoralis* has broader tarsi, an adaptation for climbing vegetation (Lott and Anderson, 2011). It is not possible to say whether the disturbed plots were a more suitable environment for these visual predators, or if the reduced density of vegetation increased pitfall trap catches (Thomas *et al.*, 2006).

4.4.5.3 Assessment of emergent groups

There was some similarity in emergent carabid and staphylinid morphological groups. Larger species were consistently associated with greater vegetation cover. Both families contained emergent groups with longer metatrochanters, although unlike the carabids, the staphylinids were associated with new scrapes. This was possibly an artefact of small sample size for staphylinid

group S5. A distinct group of staphylinids with large eyes emerged, but similar carabids including *Elaphrus riparius* and *Notiophilus* species were split among different clusters. Groups with relatively longer antennae were found in all treatments, and for different body sizes. The antennae are probably important for prey detection and habitat selection, with documentation of abundant sensilla for both carabids and staphylinids (Skilbeck and Anderson, 1996; Merivee *et al.*, 2002). Emergent groups defined by leg morphology in both families favoured scraped plots, although the carabids had longer femora, compared with shorter hind tarsi for the staphylinids. The lack of ecomorphological studies makes understanding this difficult. Where information is not available, the emergent groups could suggest potential traits to investigate for functional relevance in future (Langlands *et al.*, 2011).

There has been considerable interest in community ecology to establish whether different taxonomic groups show similar trait responses to environmental pressures (Langlands *et al.*, 2011) and potentially to characterise priority groups for conservation action (Spake *et al.*, 2016). The functional trait approach could also be used to predict responses to different management options, independent from taxonomic composition (Fountain-Jones *et al.*, 2015). In the present research, carabids and staphylinids grouped by morphological traits were associated with specific treatments, although beyond an association of larger species with less disturbed habitats, trait responses were not always shared. This is consistent with other studies (Lambeets *et al.*, 2009) and suggests difficulty in generalising trait impacts of disturbance, even between two primarily epigeal beetle families. It was difficult to characterise groups of species for conservation action. Twelve of the stenotopic early successional carabids defined in this study were measured, and these appeared in five of the six morphological groups. It is possible that smaller groups would have had greater predictive power but these often contained too little data for statistical analysis (Langlands *et al.*, 2011).

4.4.6 Implications for conservation management

This study demonstrated immediate benefits of scraping for stenotopic beetles, but these were restricted for specialist spiders. Less intensive disturbance caused limited shifts in composition of stenotopic beetles, while spider abundance increased at disturbed and control plots. From an ecological

perspective, scraping to mineral substrate can be recommended for habitat management where perennial grasses dominate. Sod-cutting was similarly preferred for heathland restoration at sites dominated by *Molinia caerulea* (Niemeyer *et al.*, 2007), although the expense of this type of management is problematic (Müller, 2004). Reduced abundance of predatory and zoophagous species after scraping emphasised the need to apply treatments at a scale to maintain the interest associated with established grassland. In addition to ecological justification or economic cost, at some sites the social acceptability of management might need consideration. Removing turf for restoration was viewed negatively by visitors to a Dutch heathland, in comparison with less intensive management (Müller, 2004).

While results in the present research were specific to Canvey Wick, these add to evidence from better-studied xeric grasslands and heathland (Pedley *et al.*, 2013; Olsson *et al.*, 2014). There were differences for longevity of treatments, particularly shorter persistence of bare ground with less intensive disturbance. This probably resulted from the hydric conditions, and was consistent with findings in Chapter 2, from restoration management at Canvey Wick in 2015. The current study could not address the frequency required for disturbance, due to the single year available for monitoring after treatments were applied. There was an apparent benefit of the earliest stages of succession for stenotopic invertebrates, but the stability of these communities was not established. In future, it will be necessary to determine whether assemblages at disturbed plots converge with control plots, or follow a different ecological trajectory. This should also be compared with succession at scraped plots created in consecutive years. In the absence of this information, both techniques could be included as prescriptions to create a heterogenous disturbance regime (Warren *et al.*, 2007).

Chapter 5 – Aggregate mound habitat creation

5.1 Introduction

Open Mosaic Habitat on Previously Developed Land is a priority habitat under Section 41 of the (Natural Environment and Rural Communities Act (2006). These are included in the duty of every public authority to have regard for the conservation of biodiversity. There is a statutory requirement for no net loss of priority habitats (Joint Nature Conservation Committee and Defra, 2012), although in practice outcomes are unclear. 51% of 198 brownfields in the Thames Gateway of medium or high importance for invertebrates had an outstanding planning permission in 2012, or were partly or fully redeveloped (Robins *et al.*, 2013). Compensatory habitat creation is required where priority habitats are lost during development (Connop *et al.*, 2016). This work is frequently *ad-hoc*, mainly in response to regulatory compliance and based on ecological intuition (Connop, 2012). The few published brownfield habitat creation projects lack replication, or are described in general terms, with little information about outcomes or management legacy (Harrison and Davies, 2002; Connop *et al.*, 2014). This makes it difficult to develop theoretical principles that would allow the transfer of methodologies from one situation to another (Hobbs and Norton, 1996).

In addition to habitat loss, fragmentation has potentially important consequences for invertebrate conservation. Isolated heathlands contained fewer stenotopic carabids (Turin and Den Boer, 1988; de Vries *et al.*, 1996). Larger areas of intensively managed land also limited community reorganisation of cold-adapted birds and butterflies (Oliver *et al.*, 2017). Research about ecological connectivity for invertebrates has focused on corridors at small spatial scales, between 2 m and 500 m (Haddad, 1999; Collinge, 2000; Berggren *et al.*, 2002; Baum *et al.*, 2004; Fried *et al.*, 2005; Haddad and Tewksbury, 2005; Tewksbury *et al.*, 2002; Haddad *et al.*, 2003; Eggers *et al.*, 2010). There is conflicting evidence from correlative studies in urban environments. Corridors were considered crucial for enhancing spider, carabid, and staphylinid biodiversity (Vergnes *et al.*, 2012), or subordinate to local habitat quality for carabids and butterflies (Angold *et al.*, 2006). For species that do not require continuous physical habitat connections, brownfield habitat creation could support metapopulation dynamics in the landscape matrix

(Lawton *et al.*, 2010). This level of understanding is very challenging and first requires reliable experimental information about changes in species richness and community structure caused by habitat creation.

Bare ground is a key component of Open Mosaic Habitat (Maddock, 2011). In other ecosystems, this is created by natural processes including fire (Grundel *et al.*, 2010), rabbit activity (Bakker and Olff, 2003), and grazing by large mammals (García *et al.*, 2011). Recently there have also been efforts to emulate natural disturbance regimes through conservation management (Warren and Büttner, 2014). A critical difference in Open Mosaic Habitat is that soil has been removed, severely modified, or covered with extraneous substrates (Maddock, 2011). Where it is not possible to disturb local soils, imported substrate could be formed into mounds. In restoration ecology, this frequently mimics natural processes, such as mammals burrowing in grassland (Umbanhowar, 1992), or soil displaced by a fallen tree (Cornett *et al.*, 1997). Mounds or ridges have been created in many ecosystems, including forests (Simmons *et al.*, 2011), wetlands (Rossell *et al.*, 2009; Bruland and Richardson, 2005; Moser *et al.*, 2007), post-industrial sites (Shaw, 1996) and restored landfills (Ewing, 2002; Biederman and Whisenant, 2011; Hough-Snee *et al.*, 2011). These studies mostly concern the establishment, diversity and survival of vegetation, and potential benefits for invertebrates are rarely considered.

The extent to which mounds promote invertebrate diversity includes characteristics of substrate and physical form. Bare or partially vegetated ground is required by some species to excavate nest burrows, including bees (Potts and Willmer, 1998), wasps (Srba and Heneberg, 2012) and beetles (Lückmann and Assmann, 2006). The proportions of sand, silt and clay in soil can determine nest initiation (Cane, 1992). Activity of many invertebrates depends on ambient temperatures (Honek, 1988) and xerothermic species may particularly benefit from bare ground. These warm up by basking (heliothermy), pressing against a warm surface (thigmothermy), or both strategies combined (Key, 2000). Afternoon soil temperatures were 8°C higher in bare mineral soil, compared with plots covered by litter and vegetation (Balisky and Burton, 1995). This can also influence oviposition and larval development. *Hesperia comma* in the UK was found to preferentially oviposit where *Festuca ovina* was largely surrounded by bare ground (Thomas *et al.*, 1986). Other invertebrates rely on

the open visual field across bare ground, including surface and aerial predators (Cannings, 1989; Knisley and Hill, 1992).

The substrate preferences of many epigeal invertebrates are hard to define but 75% of beetle species in a study of sandpits in the Czech Republic were not found at black coal spoil heaps (Tropek *et al.*, 2012; Hodecek *et al.*, 2015; Heneberg *et al.*, 2016b). Mounds also offer a niche for seed germination. Strong interactions between host plants and phytophagous beetles were observed in restoration of mesic grassland (Woodcock *et al.*, 2008). Ruderal vegetation in the early stages of vegetation succession particularly benefitted graminivorous carabids on urban derelict brownfields in the UK (Small *et al.*, 2003). Stones and cracks in the soil may offer cover for ground-dwelling invertebrates (Blight *et al.*, 2011). There have been limited attempts to examine invertebrate preferences for substrate size. Flatter species of *Bembidion* (Carabidae) were associated with gravel or stone shores, compared with more convex species at sites with sand, silt or clay (Andersen, 1985). The effects of substrate size on colonisation by benthic invertebrates have also been examined in aquatic ecosystems (Wise and Molles Jr, 1979).

Mound physical form creates variation in soil humidity and microclimate. Mound tops were more freely drained, with lower summer soil moisture compared with plots on adjacent level ground (Hough-Snee *et al.*, 2011). There were also differences between north and south-facing slopes in arid environments (Katra *et al.*, 2007). Thermal gain in the UK was maximised on south-facing slopes (Potts and Willmer, 1997) and there was 9ºC variation in ground temperature across a 23 cm high anthill, which was warmer on the south and east slopes and cooler at the base on the north and west sides (Morris *et al.*, 1994). Drier microclimate and reduced soil humidity have been interpreted as the main drivers increasing presence of early successional stenotopic species in grey dunes (Schirmel and Buchholz, 2011). Temperature can have an important effect on the development of invertebrate larvae, as co-foundress groups were larger and broods developed faster for *Polistes fuscatus* workers in sun-warmed nests compared with cooler sites (Jeanne and Morgan, 1992).

Evidence suggests that substrate addition is potentially valuable to create small-scale environmental heterogeneity. Research is required to assess the benefit for invertebrates. In this study, 5-tonne aggregate mounds were constructed in

unmanaged grasslands, with substrates contrasting to the local soils. The development of invertebrate communities on the mounds was monitored for two years. It was expected that invertebrate communities would be influenced by substrate type, and that suitability of aggregate mounds as a habitat for invertebrates would change over time. To explore this, the following questions were asked: (1) Does mound material affect colonisation by nesting aculeate Hymenoptera? (2) Does mound material affect occurrence of stenotopic early successional beetles and spiders? (3) Does mound material affect functional group structure of beetles and spiders? (4) How does habitat structure on mounds develop in two years following construction?

5.2 Methods

5.2.1 Site description

Three sites were available within low-lying unmanaged grasslands alongside the Thames Estuary. The westernmost site was adjacent to grazing marshes at Rainham (51°29'33"N, 0°13'50"W, elevation 1 m, Figure 5.1). The easternmost site was adjacent to grazing marshes at Hadleigh Park (51°32'19"N, 0°36'03"W, elevation 2 m, Figure 5.2). Between these, the third site was near grazing marshes at Fobbing, and an active landfill at Pitsea (51°32'46"N, 0°30'28"W, elevation 5 m, Figure 5.3). The exact site histories are unknown, though during this survey all the grasslands were unmanaged. These were of low botanical diversity, with *Arrhenatherum elatius*, *Dactylis glomerata*, *Cirsium arvense*, *Anthriscus sylvestris* and *Heracleum sphondylium* predominant in the sward, and scattered *Crataegus monogyna* scrub.

5.2.2 Experimental design

The statutory minimum area for recognition of Open Mosaic Habitat is 0.25 ha, although this is an administrative rather than ecological definition. Smaller areas could also have value, and be used to develop an initial understanding of Open Mosaic Habitat creation for invertebrates. These could also interact with other habitat structures, forming larger areas of mosaic habitat. There were two aggregates in the experiment. The first was quarry-dug Thanet sand, which has fine particles with little variation in size (63 μm – 1.18 mm) (Figure 5.4). The Thanet Sand Formation is a Palaeogene deposit which through glacial, periglacial and fluvial processes has been associated with exposures of fine-

Figure 5.1 Location of experimental area for aggregate mound creation at Rainham Marshes, Essex. Basemap source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS user community.

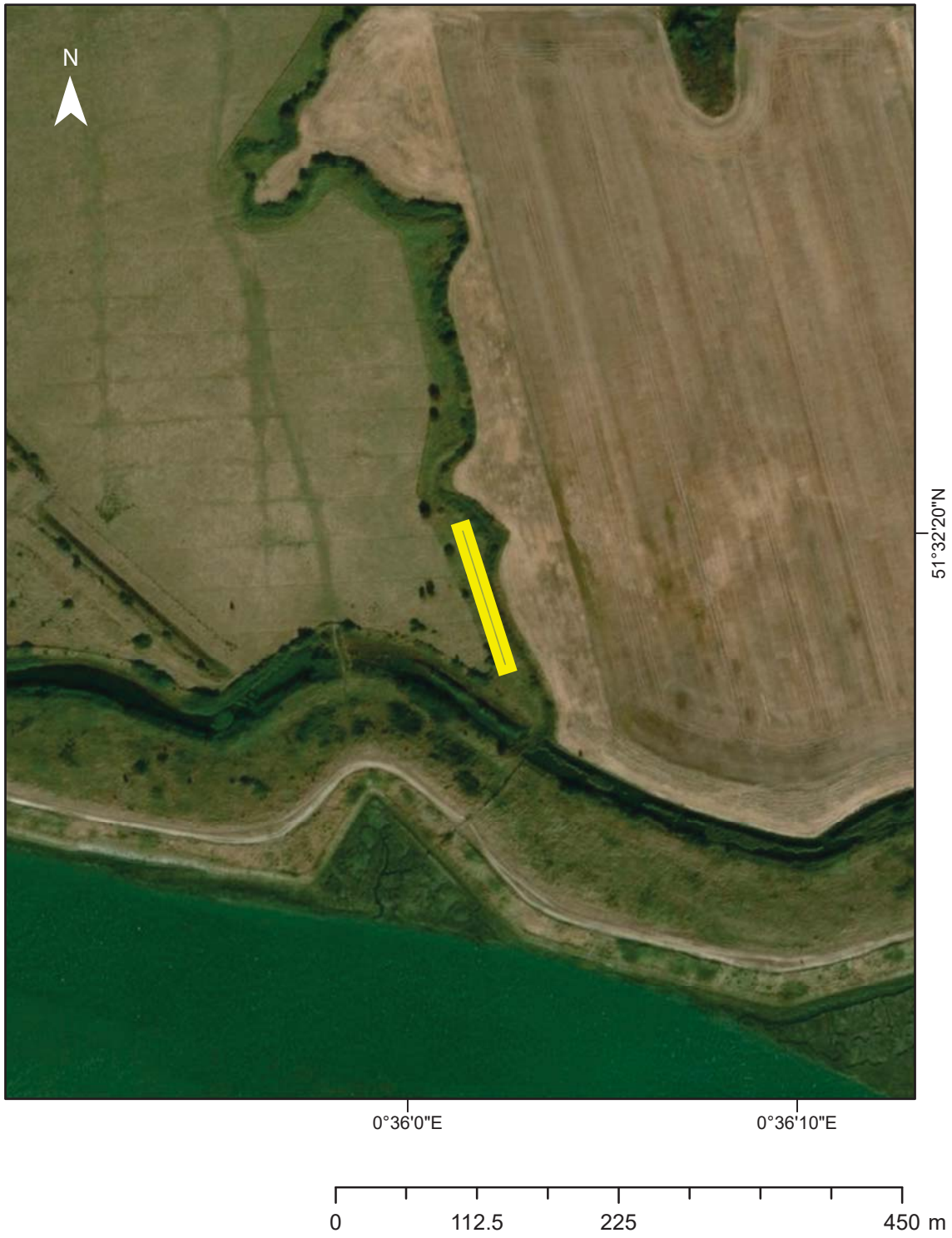

Figure 5.2 Location of experimental area for aggregate mound creation at Hadleigh Park, Essex. Basemap source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS user community.

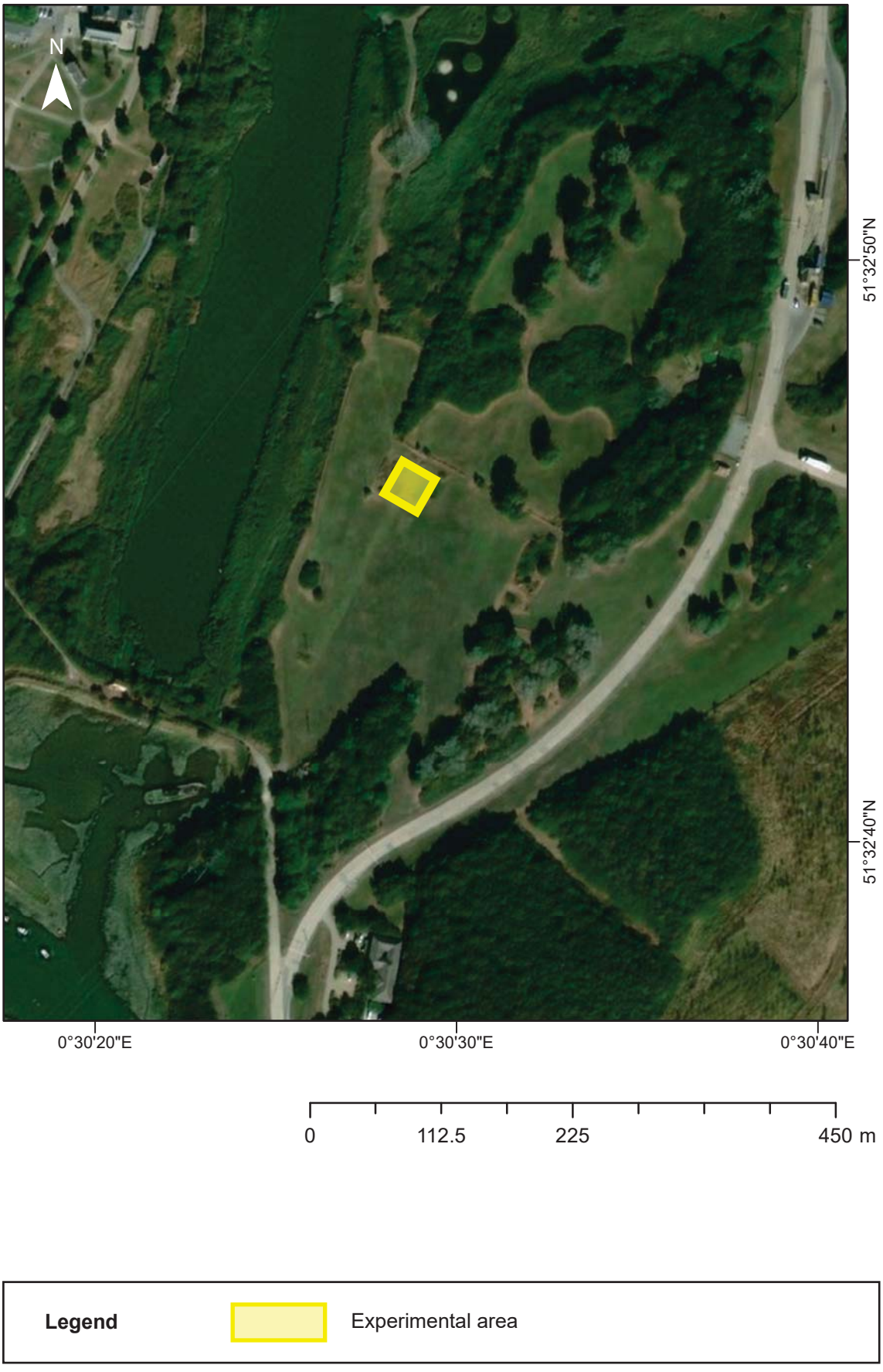

Figure 5.3 Location of experimental area for aggregate mound creation at Pitsea Landfill, Essex. Basemap source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS user community.

grained substrate in Essex and London. Historically river flood-plains were likely a major habitat for species-rich early successional invertebrate assemblages (Bogusch *et al.*, 2016), although floodplain deposits are now typically only exposed by small-scale slope failures.

In recognition of the potential environmental impact of sand quarrying (Lai *et al.*, 2014), the second material was recycled bulk fill from building site excavations, a mixture of sand and fine gravel, including larger stones (63 µm – 75 mm) (Figure 5.5). Bimodal or multimodal particle size distributions influence plant colonisation (Sadler *et al.*, 2011) and could potentially also affect patterns of invertebrate occurrence (Blight *et al.*, 2011). Each mound contained 5 tonnes of aggregate and was 0.90 m high, with an approximately 2 m^2 footprint. The piles were separated by 10 m and arranged in a randomised block design, including a grassland plot as a control in each block. This provided nine monitoring plots at each site, and 27 for the study in total (Figures 5.6-5.8). Due to a buried gas main at one site, it was not possible to excavate local substrate for mounds, or compare level areas of substrate amendment. The soils were marine clays and based on existing textural classifications (Cane, 1992) probably supported a limited range of ground-nesting aculeates.

5.2.3 Invertebrate sampling

5.2.3.1 Pitfall traps – beetles and spiders

At each monitoring plot, ground-active invertebrates were captured in two pitfall traps separated by 1 m. At the aggregate mounds, traps were midway between the base and top of the mound on the south side. In both 2015 and 2016, traps were open for fourteen-day periods in May, July and September. The sampling programme provided adequate seasonal coverage, including key periods for invertebrate activity in early successional habitats (Lott, 1998), and minimised potential depletion effects (Digweed *et al.*, 1995). Traps were polypropylene cups, 8.0 cm diameter and 10.5 cm deep, with 50 ml preservative (50% propylene glycol and 50% distilled water) and a drop of detergent to break the surface tension. To reduce capture of small mammals and reptiles, all traps were covered with 13 mm x 13 mm wire mesh. Trap contents were pooled to produce an overall sample per monitoring plot for each year.

Figure 5.4 Five-tonne pile of recycled bulk fill at Pitsea Landfill, Essex, April 2015. This material was a mixture of fine sand and gravel, including larger stones (63 μm – 75 mm). Initially the mound was 0.90 m high, with an approximately 2 m^2 footprint.

Figure 5.5 Five-tonne pile of Thanet sand at Pitsea Landfill, Essex, April 2015. This material was quarry-dug, consisting of finer particles with limited size variation (63 μm – 1.18 mm). Initially the mound was 0.90 m high, with an approximately 2 m^2 footprint.

Figure 5.6 Experimental design for aggregate mound creation at Rainham Marshes, Essex. Basemap source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS user community.

Figure 5.7 Experimental design for aggregate mound creation at Hadleigh Park, Essex. Basemap source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS user community.

Figure 5.8 Experimental design for aggregate mound creation at Pitsea Landfill, Essex. Basemap source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS user community.

5.2.3.2 Timed observations – aculeate Hymenoptera

Aculeate Hymenoptera were sampled in 2015 and 2016, at each site on single days in mid-May, late June and early August. This was undertaken between 1000 and 1500, when the temperature was above 18ºC and cloud cover less than 20%. Aculeates were sampled for 15 minutes at each mound and at control plots, with the order of sampling randomised among and within blocks. After this, a second set of observations was completed, with the order of sampling among blocks repeated from the first set, but re-randomised within blocks. This provided 30 minutes total sampling per monitoring plot for each visit. Aculeates excavating or provisioning nest holes, or quartering low over the plot surface, were captured and identified to species. Male aculeates were identified but do not form part of the analysis. Aculeates visiting flowers at the monitoring plots, or apparently hunting, were not captured.

This provided some indication of the relative abundance and diversity of aculeates potentially nesting at the monitoring plots, but did not definitely prove attempted breeding for all species. This was particularly difficult for kleptoparasites (*Nomada* and *Sphecodes*) which may have only inspected nest holes. Stem nesting aculeates were also not considered. For species that have been recorded using plant stems and nest burrows, for example *Trypoxylon medium* (Richards, 1980), this was only recorded when using a nest hole in the substrate. There was limited time for survey, and difficulty employing emergence traps at large mounds in locations with poor access. Observations of probable nesting behaviour were a surrogate for actual breeding attempts, although standardisation of methods meant that the results were comparable across all sites. Any specimens that could be identified in the field were released back onto the monitoring plots at the end of the visit. Unidentified specimens were retained for laboratory determination.

5.2.4 Invertebrate classification

5.2.4.1 Coleoptera and Araneae

Adult Coleoptera were determined to species from the references in Table 5.1, and classified from this literature as stenotopic in early successional habitats (including heathland, dry grassland, dunes, and quarries). Identifications excluded female *Gabrius* and *Anotylus sculpturatus/mutator*, and male and

Table 5.1 Identification references for Coleoptera.

Family	Reference
Alexiidae	Freude *et al.* (1967)
Anthicidae	Telnov (2010)
Apionidae	Duff (2016)
Byrrhidae	Joy (1932)
Cantharidae	Fitton (1973)
Carabidae	Luff (2007), Duff (2012)
Cerambycidae	Duff (2016)
Chrysomelidae	Duff (2016)
Coccinellidae	Majerus *et al.* (1989)
Corylophidae	Bowstead (1999)
Cryptophagidae	Coombs and Woodroffe (1955), Lohse and Klausnitzer (1992)
Curculionidae	Duff (2016)
Dermestidae	Peacock (1993)
Dytiscidae	Foster and Friday (2011)
Elateridae	Joy (1932)
Helophoridae	Foster *et al.* (2014)
Heteroceridae	Clarke (1973)
Histeridae	Duff (2012)
Hydrophilidae	Foster *et al.* (2014)
Kateretidae	Kirk-Spriggs (1996)
Latridiidae	Freude *et al.* (1967)
Leiodidae	Cooter (1996)
Malachiidae	Joy (1932)
Nitidulidae	Audisio (1993)
Oedemeridae	Buck (1954)
Phalacridae	Thompson (1958)
Ptiliidae	Duff (2012)
Scarabaeidae	Jessop (1986)
Scirtidae	Klausnitzer (2009)
Silphidae	Duff (2012)
Silvanidae	Freude *et al.* (1967)
Staphylinidae	Hammond (1972), Freude *et al.* (1974), Lott (2009), Lott and Anderson (2011), Assing and Schülke (2012)
Tenebrionidae	Brendell (1975)
Throscidae	Lucht and Klausnitzer (1998)

female staphylinids of the *Mocyta fungi* species complex. Adult spiders were determined to species from Roberts (1985) and Roberts (1987), and classified as stenotopic in early successional habitats from Harvey *et al.* (2002). Identifications excluded female *Dicymbium nigrum*/*brevisetosum*. Immature spiders were not identified due to the difficulty of classifying most species without developed reproductive structures. Species were also classified in functional groups, as predatory (including all spiders), phytophagous (herbivorous, or at least primarily granivorous), zoophagous (predatory and scavenging), and detritivorous (feeding on detritus, including saprophagous, mycetophagous, and coprophagous species), or unknown. This information was taken from the references in Table 5.1, and additionally Cooter and Barclay (2006), Anderson *et al.* (2013), Ford *et al.* (2013) and Talarico *et al.* (2016).

5.2.4.2 Aculeate Hymenoptera

Solitary bees were determined to species from Falk (2015). Solitary wasps in the families Crabronidae and Sphecidae were determined to species from Richards (1980) and Yeo and Corbet (1995). Solitary wasps in the family Pompilidae were determined to species from Day (1988).

5.2.5 Plant species composition and habitat structure

Data on plant species composition and habitat structure were collected in July 2015 and July 2016, in nine 50 cm x 50 cm quadrats. At the mounds, quadrats were located according to the cardinal compass directions. There were four quadrats around the base of each mound, four quadrats midway between the base and top, and one quadrat at the top. At grassland plots, quadrats were arranged in three rows of three, each separated by 1 m. Percentage cover of vascular plant species in each quadrat were estimated, with identifications from Stace (2010). Further variables derived from this data were species richness of forbs, graminoids, perennials and annual plants. Percentage cover of bare ground was also estimated, and litter depth was measured to the nearest millimetre. Mound height was measured directly with a 2 m ruler and horizontal bar, at four random locations along the mound top.

5.2.6 Reference sites

Reference sites can be used to evaluate the success of habitat creation (White and Walker, 1997). It was not possible to locate mounds at brownfields in the surrounding landscape, to compare with the newly established grassland mounds. Existing mounds were a poor match for size and substrate (often earthier and including brick rubble, and no examples available with only pure sand). Mounds in East London hosted populations of the Section 41 UK Priority Species *Brachinus sclopeta* (Connop, 2012), so pitfall trapping at these was not an option.

5.2.7 Statistical analysis

Univariate analyses were completed in the statistical software R (R Core Team, 2017). Multivariate analyses were completed with the Canoco 5.0 software package (ter Braak and Šmilauer, 2014). Precise *p*-values are reported, to allow interpretation of *p* as a measure of evidence, and to facilitate meta-analysis (Cumming and Calin-Jageman, 2017).

5.2.7.1 Species accumulation curves

To assess whether sampling effectively captured assemblage composition for each taxonomic group, rarefaction curves were constructed (Chao *et al.*, 2014) using the iNext package (Hsieh *et al.*, 2016).

5.2.7.2 Generalized linear mixed models

Abundance and species richness of stenotopic and generalist beetles and spiders were compared between treatments with generalized linear mixed models (GLMMs). This approach was appropriate for count data with a non-Gaussian distribution, zero probability of negative values, and heterogeneity in the mean variance relationship (Zuur *et al.*, 2009). It also allowed for potential correlation in the arrangement of treatments and controls as a randomised complete block design (Bolker *et al.*, 2009). Due to zero inflation and small sample size, datasets for stenotopic beetles and spiders were analysed in combination. Site was included as a random effect, although this was not possible for block as the number of total samples was too low to produce reliable estimates of the variance parameters. Abundance and species richness of aculeates were only compared between RBF and Thanet mounds, as no

nesting behaviour was observed in the grassland control plots. The catches from individual traps or sampling periods within each monitoring plot were pooled, to produce single totals for each species per plot year. It was not possible to analyse these separately, due to zero inflation in the subsets of data created by the species classification.

Pearson residuals from fitted models were calculated to check for overdispersion and determine the appropriate distribution for the response variable (linear, Poisson or negative binomial). These were obtained by dividing each residual by the square root of the variance (Zuur *et al.*, 2009). For model validation, residuals were plotted versus fitted values and the covariate, to assess homogeneity. Spatial autocorrelation of residuals was checked by calculation of Moran's I in the ape package (Paradis *et al.*, 2004), based on a matrix of inverse Euclidean distance weights. For all models, Moran's I was not significant ($P > 0.05$) (Table 5.3, Table 5.4). Differences among treatment means were examined by Tukey pair-wise comparisons. To adjust *p*-values, the family-wise error rate was controlled using the Holm-Šidák adjustment (Holm, 1979), previously described in section 2.6.1. Abundance of aculeate Hymenoptera was also compared for the two mound types in May, June and August 2015 and 2016. Due to zero inflation and the small sample size, this data was analysed with the Wilcoxon matched pairs test for each pair of mounds within the nine experimental blocks.

5.2.7.3 Environmental variables

GLMMs were not able to consistently predict the environmental variables across all treatment classes, due to heteroscedasticity. As a result, the variables were analysed with the Kruskal–Wallis (H) test (Kruskal and Wallis, 1952), followed by Dunn's test to report the results among multiple pairwise comparisons (Dunn, 1964). To adjust *p*-values the family-wise error rate was controlled using the Holm-Šidák adjustment (Holm, 1979) in the dunn.test package (Dinno, 2016). This is a progressive step-up procedure assuming dependence between tests, which are completed in order of increasing probability α/n-1. This was preferred to the classical Bonferroni correction, with tests compared to α/n, which gives a smaller or equal probability of rejecting false hypotheses for the same data (Holm, 1979).

5.2.7.4 Ordination

Assemblage composition for aculeate Hymenoptera and stenotopic beetles and spiders was examined with non-metric multidimensional scaling (NMDS). This was preferred due to the probability of considerable species turnover along the treatment gradient, which can cause a strong arch effect in metric ordination techniques (Prentice, 1977). Aculeate Hymenoptera were compared at RBF and Thanet mounds, as all grassland plots produced zero scores. Stenotopic species were compared at RBF mounds, Thanet mounds, and grassland controls. To produce a stable NMDS solution, samples from both years were pooled to produce a single abundance per mound. Ordinations were completed with presence/absence data, and repeated with abundance data for alternative representation of the community structure. Ordinations were based on a matrix of Jaccard dissimilarities for presence/absence data, and Bray-Curtis dissimilarities for abundance data, with a maximum of 100 random starts to search for the lowest stress solution, k = 3. Stress values were compared with guidelines suggested by Clarke (1993): <0.05 = excellent, <0.10 = good, <0.20 = usable, >0.20 = not acceptable. The contributions of substrate and site to the occurrence of aculeate Hymenoptera and stenotopic beetles and spiders were investigated with variance partitioning (Borcard *et al.*, 1992).

Constrained ordination analysis was used to explore the proportion of variation in functional group data that could be explained by environmental variables. Samples within each monitoring plot were pooled within years to compile a matrix of log-transformed abundance values. Gradient lengths were assessed by Detrended Correspondence Analysis (DCA) of this data. Axis one was short (0.5). This suggested the data showed linear response, so ordination was generated by Redundancy Analysis (RDA) (Paliy and Shankar, 2016). The environmental variables were percentage of bare ground, litter depth, and four vegetation species richness measures (forb, graminoid, annual, and perennial). RDA scaling was focused on inter-species correlations and centred by species. Substrate and site were included in the final triplot but were not used to influence the analysis. The significance of environmental variables was tested using automatic forward selection (Monte Carlo test, 500 permutations) within a split-plot design. Permutations were made within but not between whole plots that were defined by sites. To compensate for multiple testing of variables

during forward selection and reduce the chance of type I errors, the Holm correction was used where successive tests are multiplied by *q*-1 (Holm, 1979).

5.3 Results

5.3.1 Environmental variables

Forb species richness differed significantly between treatments in 2015 ($\chi^2_{[2, N=27]}$ = 12.95, P = 0.0015) (Table 5.2). This was significantly higher at RBF mounds (P_{adj} = 0.0005) and control plots (P_{adj} = 0.0499) than Thanet mounds, and marginally higher at RBF mounds than control plots (P_{adj} = 0.0506). Herbaceous plant species richness also differed significantly between treatments in 2016 ($\chi^2_{[2, N=27]}$ = 12.95, P = 0.0015). This was significantly higher at RBF mounds (P_{adj} = 0.0006) and Thanet mounds (P_{adj} = 0.0429) than control plots. There was no significant difference in overall graminoid richness for treatments in 2015 ($\chi^2_{[2, N=27]}$ = 5.84, P = 0.040), although this was significantly higher in control plots compared with RBF mounds (P_{adj} = 0.0261). There was also no significant difference in graminoid richness between treatments in 2016 ($\chi^2_{[2, N=27]}$ = 0.78, P = 0.6775).

Annual plant species richness differed significantly between treatments in 2015 ($\chi^2_{[2, N=27]}$ = 10.47, P = 0.0053). This was significantly higher at RBF mounds compared with Thanet mounds (P_{adj} = 0.0002) and control plots (P_{adj} = 0.0453). Annual plant species richness also differed significantly between treatments in 2016 ($\chi^2_{[2, N=27]}$ = 7.55, P = 0.0133). This was significantly higher at RBF mounds (P_{adj} = 0.0133) and Thanet mounds (P_{adj} = 0.0412) than control plots.

Perennial plant species richness did not differ significantly between treatments in 2015 ($\chi^2_{[2, N=27]}$ = 2.61, P = 0.2715) or 2016 ($\chi^2_{[2, N=27]}$ = 1.10, P = 0.5756). Litter depth differed significantly between treatments in 2015 ($\chi^2_{[2, N=27]}$ = 18.32, P < 0.0001). This was significantly greater in control plots than Thanet or RBF mounds (P_{adj} < 0.0001 in both cases). Litter depth also differed significantly between treatments in 2016 ($\chi^2_{[2, N=27]}$ = 20.43, P < 0.0001). This was significantly greater at control plots than RBF mounds (P_{adj} < 0.0001) or Thanet mounds (P_{adj} = 0.0063), and significantly greater at Thanet mounds than RBF mounds (P_{adj} = 0.0399).

Table 5.2 Medians (minimum-maximum) and results of Kruskal–Wallis (H) test, comparing environmental characteristics at aggregate mounds experiment in south Essex, 2015 and 2016. Treatments were 5 tonne piles of recycled bulk fill (n=9), Thanet sand (n=9), and unmanaged grassland controls (n=9). Vegetation species and structural variables were surveyed in July in both years. Post-hoc tests were Tukey pairwise comparisons after Holm-Šidák adjustment, treatments with shared superscript a-c (ranked highest to lowest) did not differ significantly ($P > 0.05$).

Year	Variable	Control	RBF	Thanet	Kruskal-Wallis$_{[2, N=27]}$
2015	Forb species richness	5 (1–8) [a]	7 (4–9) [a]	2 (2–4) [b]	$H = 12.95$
	Graminoid species richness	6 (2–10) [a]	2 (1–8) [ab]	3 (2–5) [b]	$H = 5.84$
	Annual species richness	1 (0–2) [b]	2 (1–6) [a]	0 (0–2) [b]	$H = 10.47$
	Perennial species richness	11 (3–15) [a]	5 (4–14) [a]	5 (3–10) [a]	$H = 2.61$
	Litter depth (cm)	3.1 (1.3–8.0) [a]	0.5 (0.2–1.1) [b]	0.3 (0.02–0.6) [b]	$H = 18.32$
	Bare ground (%)	0.0 (0.0–2.8) [b]	74.4 (50.6–95) [a]	82.8 (47.2–90.3) [a]	$H = 17.59$
2016	Forb species richness	3 (2–6) [b]	6 (5–12) [a]	5 (3–9) [a]	$H = 12.63$
	Graminoid species richness	4 (2–11) [a]	3 (2–6) [a]	4 (2–11) [a]	$H = 0.78$
	Annual species richness	1 (0–5) [b]	3 (2–10) [a]	2 (2–7) [a]	$H = 7.55$
	Perennial species richness	9 (3–14) [a]	8 (5–10) [a]	6 (3–13) [a]	$H = 1.10$
	Litter depth (cm)	3.8 (1.6–7.6) [a]	0.8 (0.5–1.1) [c]	1.1 (0.8–1.4) [b]	$H = 20.43$
	Bare ground (%)	0.0 (0.0–1.1) [b]	2.2 (1.1–44.4) [a]	1.6 (0.5–56.1) [a]	$H = 16.31$

Percentage cover of bare ground differed significantly between treatments in 2015 ($\chi^2_{[2, N=27]}$ = 17.59, P < 0.0001). This was significantly higher at Thanet mounds (P_{adj} = 0.0003) and RBF mounds (P_{adj} = 0.0004) than control plots. Percentage cover of bare ground also differed significantly between treatments in 2016 ($\chi^2_{[2, N=27]}$ = 16.31, P = 0.0002). This was significantly higher at RBF mounds (P_{adj} = 0.0003) and Thanet mounds (P_{adj} = 0.0013) than control plots. The average size of the mounds also decreased over time. At the end of the first year, the average height of RBF mounds was 84.7 cm (–5.9% of original height). The average height of Thanet mounds was 81.6 cm (–9.3%). At the end of the second year, the average height of RBF mounds was 82.5 cm (–8.3% of original height). The average height of Thanet mounds was 73.5 cm (–18.3%). One side of a Thanet mound at Hadleigh was partially trampled by a cow before the first monitoring round in 2015. This was reshaped, and after repair of a fence there was no further disturbance at this site.

5.3.2 Beetles and spiders - overall summary

During the study 14,772 invertebrates of 499 species were identified, comprising 11,035 beetles and 3,737 spiders. A further 12 invertebrates probably representing two beetle and one spider species could not be identified from female specimens. Male specimens of both beetle species likely to be present were identified, although only one female *Dicymbium nigrum*/*brevisetosum* was recorded. In the staphylinid genus *Mocyta*, 72 male and female specimens were not identified because of taxonomic uncertainty. These, and female specimens for the other three species, are excluded from subsequent analysis. A sample-based rarefaction curve estimated 99 (SE ± 10) spider species likely to be captured in pitfall traps in the sampled habitats in 2015 (Figure 5.9). Eighty-two species captured represent 82.8% of this estimated fauna. From an estimated 376 species of beetle (SE ± 31), 70.7% or 266 species were captured (Figure 5.9). A sample-based rarefaction curve estimated 100 (SE ± 11) spider species likely to be captured in pitfall traps in the sampled habitats in 2016 (Figure 5.10). Seventy-eight species captured represent 78% of this estimated fauna. From an estimated 526 species of beetle (SE ± 11), 62.7% or 330 species were captured (Figure 5.10).

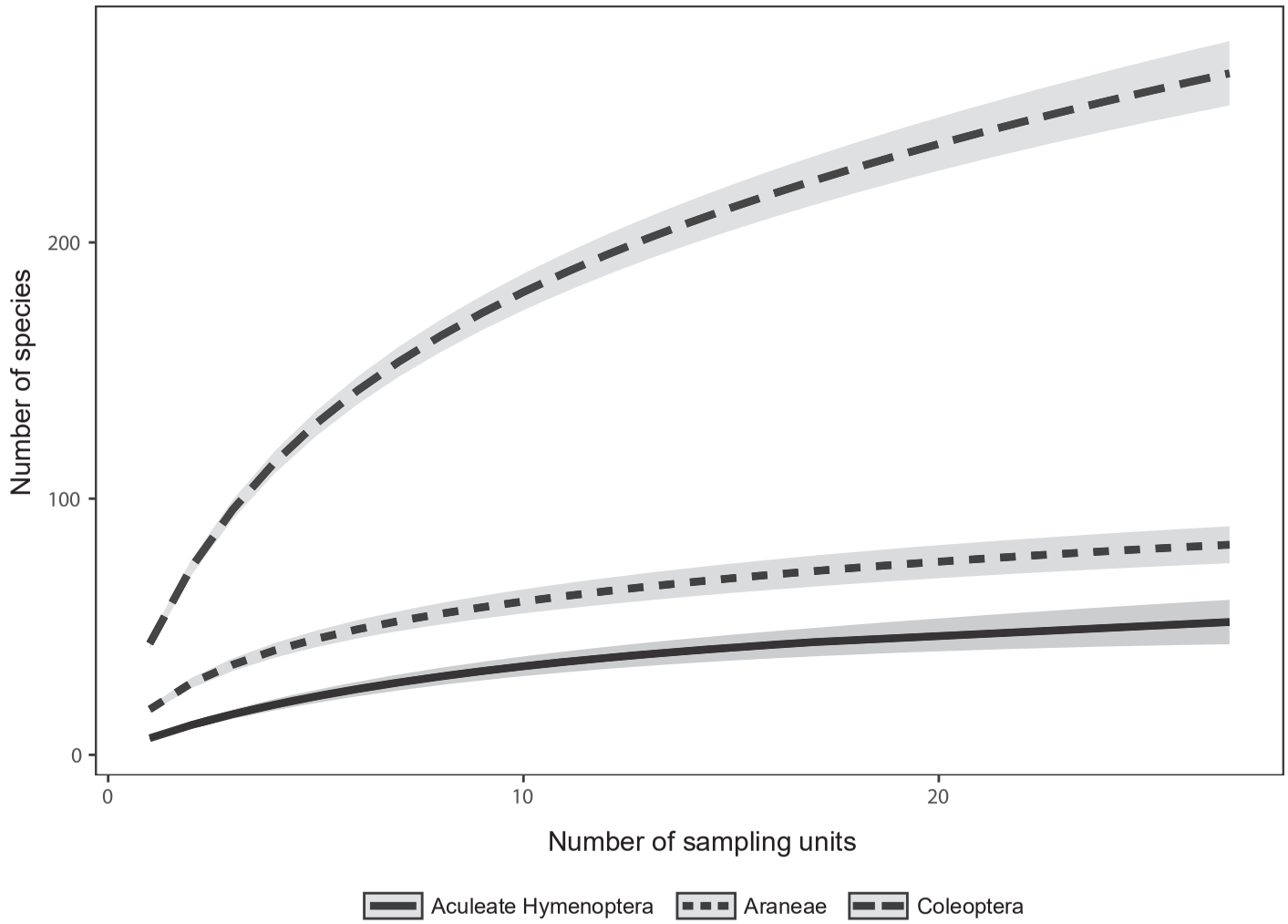

Figure 5.9 Sample-based rarefaction curves of species richness for aculeate Hymenoptera, Araneae and Coleoptera, based on species likely to be captured by timed searches (aculeates), or pitfall traps (Araneae and Coleoptera) at 5-tonne aggregate mounds and grassland controls, in south Essex in 2015 (n=27). Shaded areas represent 95% confidence intervals. Curves become more shallow approaching the asymptotes of the estimated faunas.

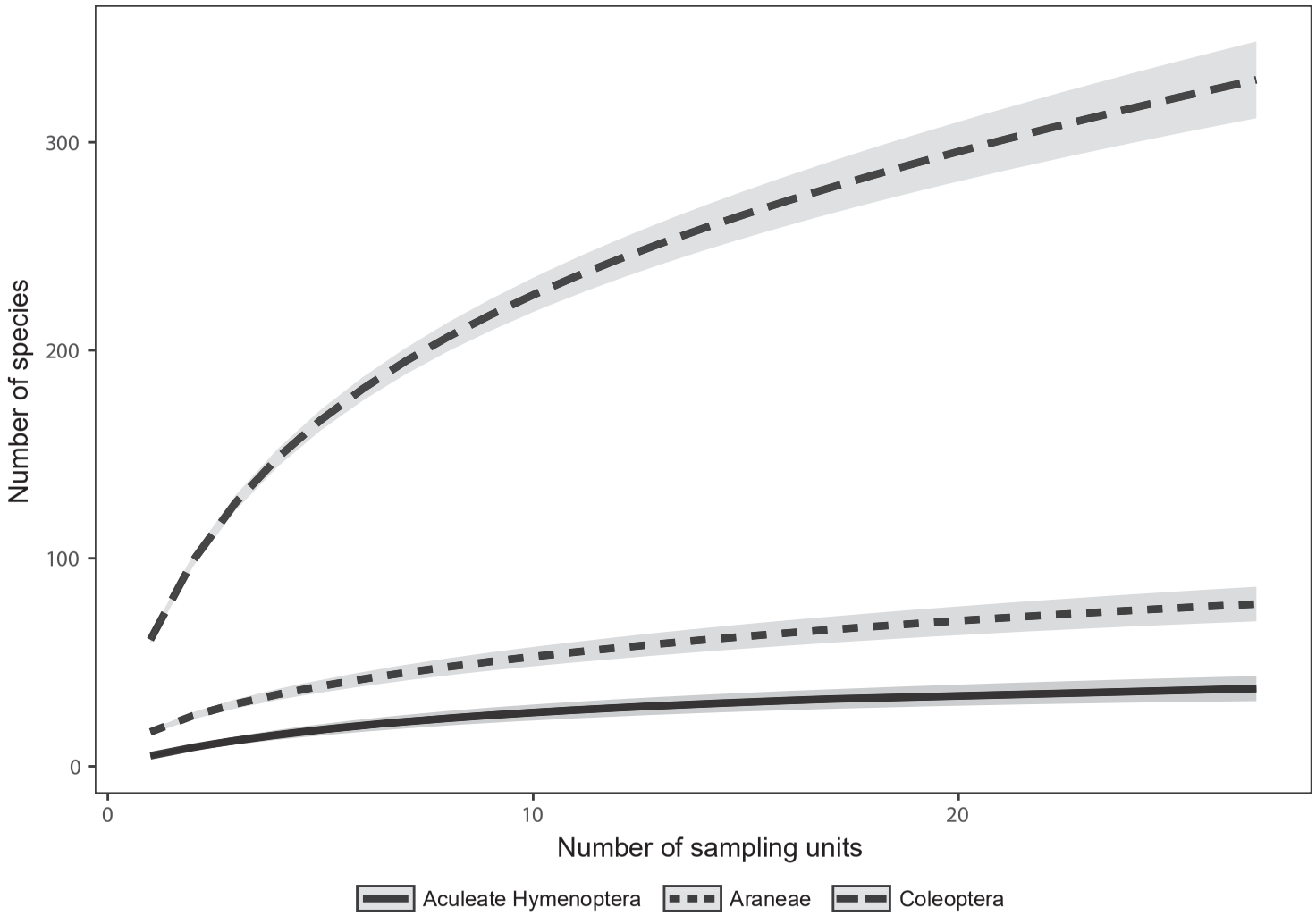

Figure 5.10 Sample-based rarefaction curves of species richness for aculeate Hymenoptera, Araneae and Coleoptera, based on species likely to be captured by timed searches (aculeates), or pitfall traps (Araneae and Coleoptera) at 5-tonne aggregate mounds and grassland controls, in south Essex in 2016 (n=27). Shaded areas represent 95% confidence intervals. Curves become more shallow approaching the asymptotes of the estimated faunas.

5.3.3 Beetle summary - treatments

Totals for beetles trapped at the monitoring plots were 3,987 in 2015 and 7,060 in 2016 (Appendix 9.5, Table A9.5.1, page 334). The number of individuals increased at the control plots from 1,833 in 2015, to 2,425 in 2016. From 171 species trapped in 2015, the most abundant were the predator *Drusilla canaliculata* (29%), the predator *Ocypus olens* (6%) and the phytophage *Agriotes sputator* (6%). There were 106 species represented by three or fewer individuals, forming 8.6% of the catch. From 195 species trapped in 2016, the most abundant were *Drusilla canaliculata* (30%), the predator *Pterostichus madidus* (7%), *Ocypus olens* and *Agriotes sputator* (5% in both cases). There were 121 species represented by three or fewer individuals, forming 4.8% of the catch.

The number of individuals increased at the RBF mounds from 1,363 in 2015, to 2,520 in 2016. From 168 species trapped at the RBF mounds in 2015, the most abundant were *Ocypus olens* (7%), *Pterostichus madidus* (6%) and *Drusilla canaliculata* (5%). There were 102 species represented by three or fewer individuals, forming 10.9% of the catch. From 212 species trapped at the RBF mounds in 2016, the most abundant were phytophagous *Longitarsus dorsalis* (8%), *Pterostichus madidus* (8%) and phytophagous *Phyllotreta atra* (7%). There were 112 species represented by three or fewer individuals, forming 6.5% of the catch.

The number of individuals increased at the Thanet mounds from 791 in 2015, to 1,912 in 2016. From 146 species trapped at the Thanet mounds in 2015, the most abundant were the predator *Tachyporus nitidulus* (7%), *Drusilla canaliculata* (6%) and *Pterostichus madidus* (5%). There were 97 species represented by three or fewer individuals, forming 19.6% of the catch. From 228 species trapped at the Thanet mounds in 2016, the most abundant were Drusilla canaliculata (8%), *Pterostichus madidus* (7%) and *Tachyporus nitidulus* (7%). There were 144 species represented by three or fewer individuals, forming 11.3% of the catch.

5.3.4 Spider summary - treatments

Totals for spiders trapped at the monitoring plots were 1,682 in 2015 and 2,055 in 2016 (Appendix 9.5, Table A9.5.2, page 334). The number of individuals

increased at the control plots from 928 in 2015, to 1,113 in 2016. From 56 species trapped in 2015, the most abundant were *Alopecosa pulverulenta* (21%), *Pardosa prativaga* (19%) and *Pardosa palustris* (12%). There were 32 species represented by three or fewer individuals, forming 5.0% of the catch. From 53 species trapped in 2016, the most abundant were *Alopecosa pulverulenta* (36%), *Pardosa prativaga* (11%) and *Pardosa pullata* (10%). There were 33 species represented by three or fewer individuals, forming 5.1% of the catch.

The number of individuals decreased at the RBF mounds from 481 in 2015, to 442 in 2016. From 55 species trapped at the RBF mounds in 2015, the most abundant were *Pardosa prativaga* (19%), *Pardosa nigriceps* (11%) and *Alopecosa pulverulenta* (8%). There were 28 species represented by three or fewer individuals, forming 10.0% of the catch. From 47 species trapped at the RBF mounds in 2016, the most abundant were *Alopecosa pulverulenta* (21%), *Pardosa prativaga* (13%) and *Pardosa nigriceps* (10%). There were 28 species represented by three or fewer individuals, forming 8.4% of the catch.

The number of individuals decreased at the Thanet mounds from 791 in 2015, to 500 in 2016. From 60 species trapped at the Thanet mounds in 2015, the most abundant were *Pardosa nigriceps* (18%), *Pardosa prativaga* (14%) and *Alopecosa pulverulenta* (7%). There were 41 species represented by three or fewer individuals, forming 23.8% of the catch. From 47 species trapped at the Thanet mounds in 2016, the most abundant were *Alopecosa pulverulenta* (27%), *Pardosa nigriceps* (14%) and *Pardosa prativaga* (10%). There were 30 species represented by three or fewer individuals, forming 8.4% of the catch.

5.3.5 Stenotopic beetles and spiders

Stenotopic species differed in abundance ($\chi^2_{[2, N=27]} = 15.79$, $P = 0.0003$) between treatments in 2015, but did not differ in richness ($\chi^2_{[2, N=27]} = 5.29$, $P = 0.071$) (Table 5.3). Abundance of stenotopic species was significantly higher at RBF mounds than Thanet mounds ($P_{adj} = 0.0003$). Stenotopic species differed in abundance ($\chi^2_{[2, N=27]} = 21.39$, $P < 0.0001$) in 2016. Abundance of stenotopic species was significantly higher at RBF mounds than control plots ($P_{adj} < 0.0001$) or Thanet mounds ($P_{adj} = 0.0073$). The global statistic for richness of stenotopic species was significant in 2016 ($\chi^2_{[2, N=27]} = 6.02$, $P = 0.0494$).

Table 5.3 Mean ± standard error for species richness and abundance of stenotopic beetles and spiders, and aculeate Hymenoptera, in aggregate mound experiment, 2015-2016. Beetle and spider samples were pooled from two pitfall traps within each monitoring plot (n=27), open for 14 days in May, July, and September. Aculeate Hymenoptera samples were pooled from 30 minutes hand netting per plot, during three visits in May, June, and August, 2015-2016. Generalized linear mixed models (χ^2, F) compare 5-tonne piles of recycled bulk fill or Thanet sand, and unmanaged grassland controls. There was zero data for aculeate Hymenoptera from the unmanaged grassland plots, and these were not included in models. Post-hoc tests show Tukey pairwise comparisons after Holm-Šidák adjustment, treatments with shared superscript a-b (ranked highest to lowest) did not differ significantly ($P > 0.05$). Moran's I was calculated to test for spatial correlation in GLMM residuals, based on a matrix of inverse Euclidean distance weights (significance level $P < 0.05$).

Invertebrate group	Variable	Year	Control	RBF	Thanet	$GLMM_{[2, N=27]}$	Moran's I_{GLMM}
Coleoptera + Araneae	Abundance	2015	13.3 ± 2.5 ^{ab}	19.3 ± 3.1 ^{a}	8.7 ± 1.3 ^{b}	$\chi^2 = 15.79$	0.1178
		2016	12.9 ± 2.3 ^{b}	30.2 ± 4.4 ^{a}	17.1 ± 1.9 ^{b}	$\chi^2 = 21.39$	0.3069
	Species richness	2015	7.3 ± 1.1 ^{a}	9.7 ± 0.9 ^{a}	6.9 ± 0.9 ^{a}	$F = 5.29$	0.8314
		2016	6.9 ± 1.1 ^{a}	9.5 ± 0.9 ^{a}	10.0 ± 0.9 ^{a}	$F = 6.01$	0.9090
Aculeate Hymenoptera	Abundance	2015	0.0 ± 0.0	8.1 ± 2.0 ^{b}	24.9 ± 3.9 ^{a}	$\chi^2 = 80.48$	0.7204
		2016	0.0 ± 0.0	9.2 ± 2.5 ^{b}	16.4 ± 4.5 ^{a}	$\chi^2 = 3.52$	0.9182
	Species richness	2015	0.0 ± 0.0	6.3 ± 0.5 ^{b}	12.0 ± 1.9 ^{a}	$F = 16.03$	0.4167
		2016	0.0 ± 0.0	5.3 ± 1.1 ^{b}	8.9 ± 2.0 ^{a}	$\chi^2 = 8.09$	0.9067

Species richness was higher at Thanet mounds (P_{adj} = 0.0718) and RBF mounds (P_{adj} = 0.231) than control plots, although differences between treatments were not significant.

There were 74 species of stenotopic beetles and spiders recorded at the aggregate mounds and grassland control plots. Fifty-one species occurred at more than one site. Substrate and site explained 20% of variation in stenotopic species occurrence (F=2.66, P = 0.001). As conditional effects, site explained the largest proportion of variation at 17% (F=3.62, P = 0.001), substantially higher than the 5% explained by substrate (F=1.70, P = 0.002). The shared variation between substrate and site was negative at -2%, as the variables are complementary in explanatory power and explain the response variable better than the sum of the individual effects of these variables (Jongman, 1995).

NMDS ordination based on presence/absence data (Figure 5.11) showed assemblages of stenotopic beetles and spiders clearly separated by site (stress value 0.1433). Hadleigh was separated on the first axis, and Pitsea and Rainham on the second axis, although one of the grassland plots at Rainham was more similar to the mounds at Pitsea. The separation of grassland and mounds within sites was less distinct. Grassland samples were separated from RBF and Thanet mounds at Hadleigh, although the mounds were in a similar part of the ordination space. The greatest distinction at Rainham was between Thanet mounds and grassland controls, with RBF mounds at an intermediate position. At Pitsea, RBF and Thanet mounds showed greater separation, with grassland controls at an intermediate position. NMDS based on abundance (Figure 5.12) showed control plots separated from treatments at each site (stress value 0.1324). At Hadleigh treatments also were separated, whereas these overlapped at Rainham and Pitsea. In this ordination, Pitsea was separated on the first axis, and Rainham and Hadleigh on the second axis, although control plots at Hadleigh partly overlapped with Thanet mounds at Rainham.

There were 41 stenotopic species at control plots, 51 species at RBF mounds and 56 species at Thanet mounds. Five species were only recorded at grassland controls. Beetles were *Amara tibialis*, *Elaphrus riparius* and *Harpalus anxius* (Carabidae), *Neobisnius villosulus* and *Carpelimus corticinus* (Staphylinidae). One spider only recorded at grassland plots was *Drassodes*

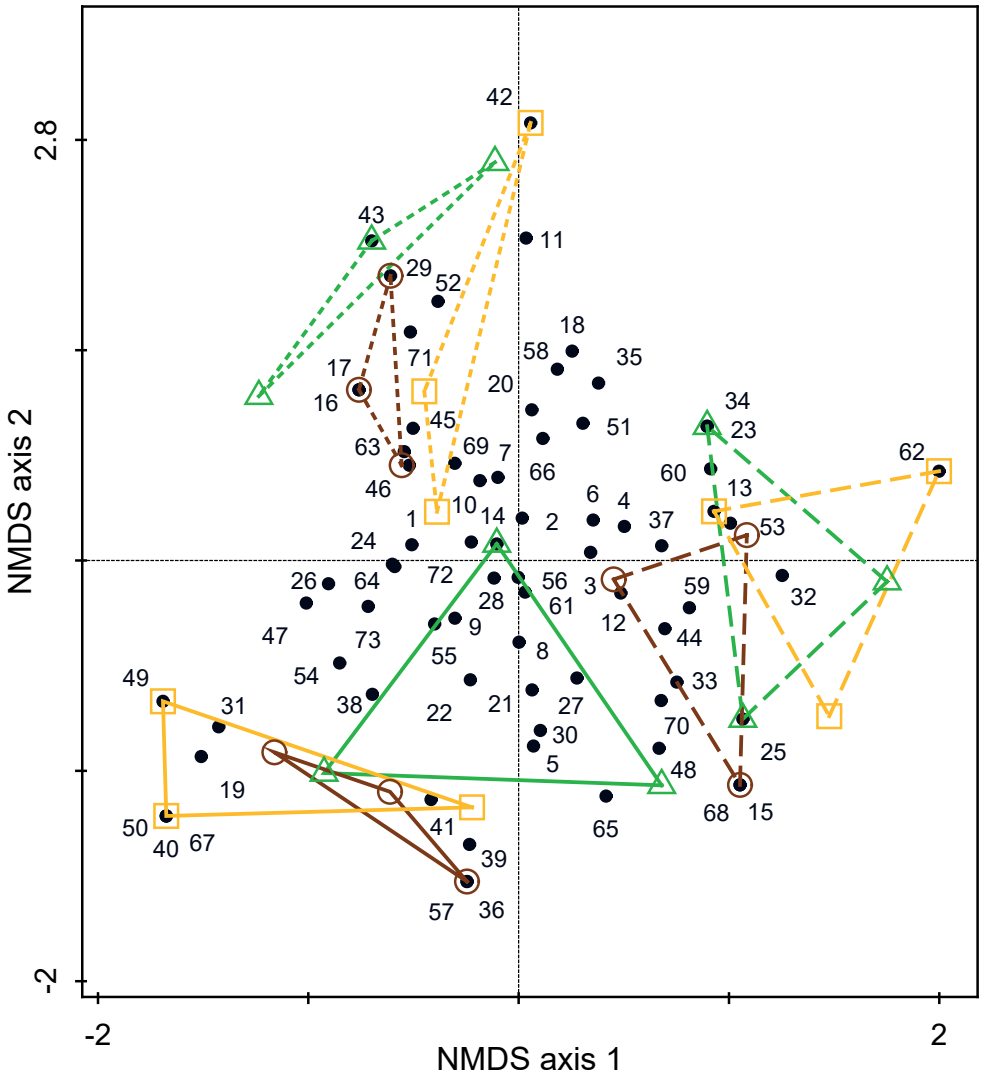

Figure 5.11 Biplot scores for stenotopic early successional beetle and spider species (●) and treatment plots, calculated from Non-metric Multidimensional Scaling of presence-absence data at south Essex aggregate mounds, 2015-2016 (stress value = 0.13994, 3 dimensions). Ordination was based on a matrix of Jaccard dissimilarities, with a maximum of 100 random starts to search for the lowest stress solution. Samples were pooled in both years from two pitfall traps per monitoring plot, open for 14 days in May, July, and September. Treatments were 5-tonne mounds of recycled bulk fill (n=9) or Thanet sand (n=9), and unmanaged grassland controls (n=9). Treatments are coded recycled bulk fill (◯), Thanet sand (□), unmanaged grassland (△). Sites are coded Hadleigh Park (- - - -), Pitsea Landfill (— —), Rainham Marshes (——). Numerical references for species names:
1 = *Amara aenea*, 2 = *Amara convexior*, 3 = *Amara eurynota*, 4 = *Amara familiaris*, 5 = *Amara tibialis*, 6 = *Anotylus inustus*, 7 = *Anthicus antherinus*, 8 = *Argenna subnigra*, 9 = *Astenus lyonessius*, 10 = *Bembidion femoratum*, 11 = *Bembidion lunulatum*, 12 = *Brachinus crepitans*, 13 = *Calathus ambiguus*, 14 = *Carpelimus corticinus*, 15 = *Carpelimus incongruus*, 16 = *Carpelimus rivularis*, 17 = *Centromerus capucinus*, 18 = *Crustulina guttata*, 19 = *Curtonotus convexiusculus*, 20 = *Drassodes cupreus*, 21 = *Drassyllus pusillus*, 22 = *Dysdera crocata*, 23 = *Elaphrus riparius*, 24 = *Haplodrassus signifer*, 25 = *Harpalus anxius*, 26 = *Harpalus attenuatus*, 27 = *Harpalus rubripes*, 28 = *Harpalus tardus*, 29 = *Helophorus rufipes*, 30 = *Hypera fuscocinerea*, 31 = *Lobrathium multipunctum*, 32 = *Melanophthalma suturalis*, 33 = *Microlestes minutulus*, 34 = *Neobisnius villosulus*, 35 = *Notiophilus substriatus*, 36 = *Omphalapion hookerorum*, 37 = *Ophonus ardosiacus*, 38 = *Ophonus azureus*, 39 = *Otiorhynchus ligneus*, 40 = *Otiorhynchus ovatus*, 41 = *Otiorhynchus raucus*, 42 = *Oxypoda brachyptera*, 43 = *Oxypoda procerula*, 44 = *Ozyptila simplex*, 45 = *Paederus littoralis*, 46 = *Panagaeus bipustulatus*, 47 = *Paradromius linearis*, 48 = *Pardosa agrestis*, 49 = *Philhygra debilis*, 50 = *Philhygra palustris*, 51 = *Philorhizus melanocephalus*, 52 = *Phrurolithus minimus*, 53 = *Platyderus depressus*, 54 = *Platystethus alutaceus*, 55 = *Platystethus nitens*, 56 = *Poecilus cupreus*, 57 = *Romualdius angustisetulus*, 58 = *Rugilus orbiculatus*, 59 = *Scybalicus oblongiusculus*, 60 = *Sibianor aurocinctus*, 61 = *Sunius propinquus*, 62 = *Tasgius ater*, 63 = *Tasgius globulifer*, 64 = *Tasgius winkleri*, 65 = *Tegenaria agrestis*, 66 = *Trachyzelotes pedestris*, 67 = *Troxochrus scabriculus*, 68 = *Walckenaeria capito*, 69 = *Xantholinus elegans*, 70 = *Xysticus kochi*, 71 = *Zelotes apricorum*, 72 = *Zelotes latreillei*, 73 = *Zodarion italicum*.

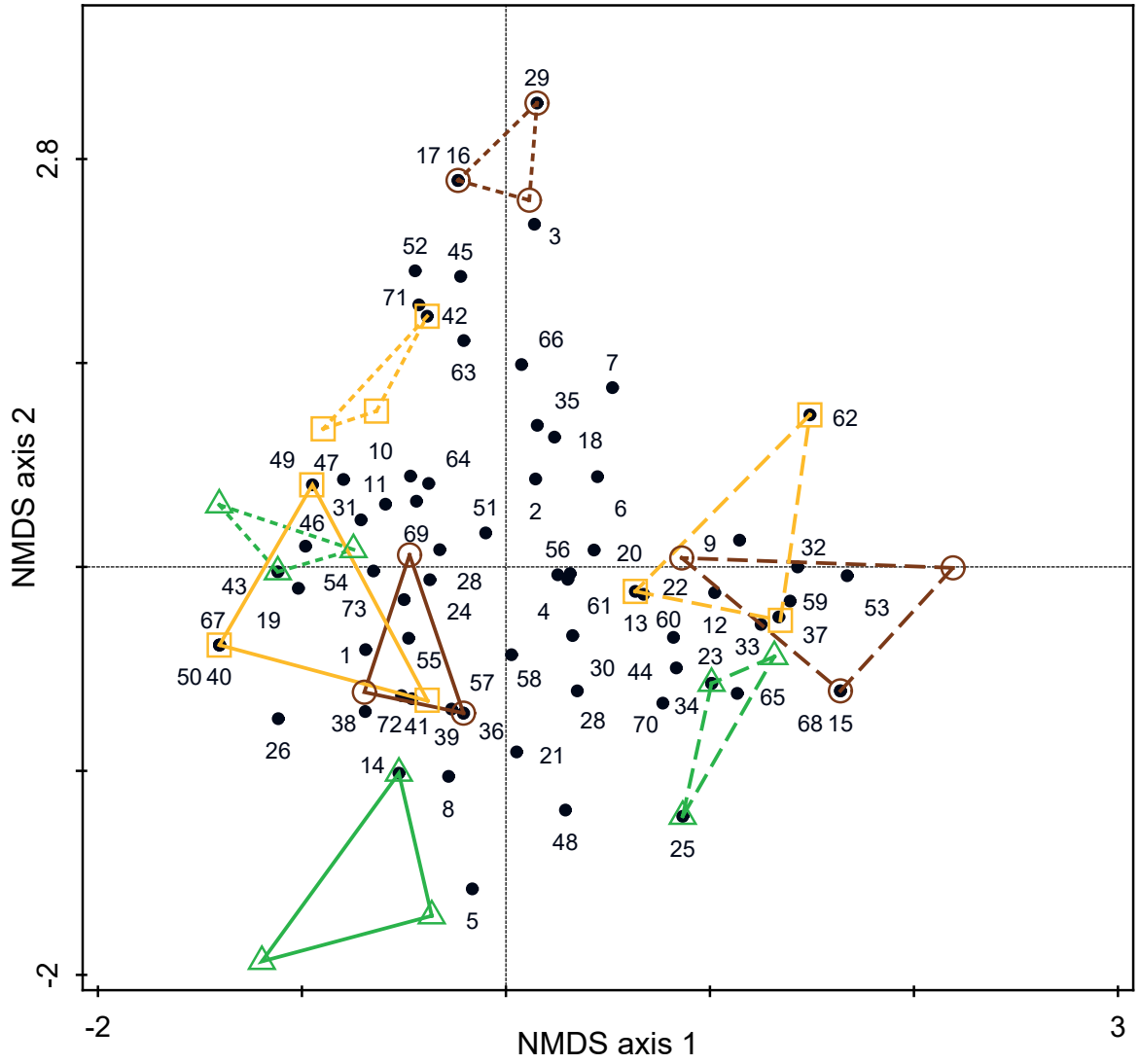

Figure 5.12 Biplot scores for stenotopic early successional beetle and spider species (●) and treatment plots, calculated from Non-metric Multidimensional Scaling of abundance data at south Essex aggregate mounds, in 2015- 2016 (stress value = 0.13228, 3 dimensions). Ordination was based on a matrix of Bray-Curtis dissimilarities, with a maximum of 100 random starts to search for the lowest stress solution. Samples were pooled in each year from two pitfall traps per monitoring plot, open for 14 days in May, July, and September. Treatments were 5-tonne mounds of recycled bulk fill (n=9) or Thanet sand (n=9), and unmanaged grassland controls (n=9). Treatments are coded recycled bulk fill (◯), Thanet sand (□), unmanaged grassland (△). Sites are coded Hadleigh Park (- - - -), Pitsea Landfill (— —), Rainham Marshes (——). Numerical references for species names:
1 = *Amara aenea*, 2 = *Amara convexior*, 3 = *Amara eurynota*, 4 = *Amara familiaris*, 5 = *Amara tibialis*, 6 = *Anotylus inustus*, 7 = *Anthicus antherinus*, 8 = *Argenna subnigra*, 9 = *Astenus lyonessius*, 10 = *Bembidion femoratum*, 11 = *Bembidion lunulatum*, 12 = *Brachinus crepitans*, 13 = *Calathus ambiguus*, 14 = *Carpelimus corticinus*, 15 = *Carpelimus incongruus*, 16 = *Carpelimus rivularis*, 17 = *Centromerus capucinus*, 18 = *Crustulina guttata*, 19 = *Curtonotus convexiusculus*, 20 = *Drassodes cupreus*, 21 = *Drassyllus pusillus*, 22 = *Dysdera crocata*, 23 = *Elaphrus riparius*, 24 = *Haplodrassus signifer*, 25 = *Harpalus anxius*, 26 = *Harpalus attenuatus*, 27 = *Harpalus rubripes*, 28 = *Harpalus tardus*, 29 = *Helophorus rufipes*, 30 = *Hypera fuscocinerea*, 31 = *Lobrathium multipunctum*, 32 = *Melanophthalma suturalis*, 33 = *Microlestes minutulus*, 34 = *Neobisnius villosulus*, 35 = *Notiophilus substriatus*, 36 = *Omphalapion hookerorum*, 37 = *Ophonus ardosiacus*, 38 = *Ophonus azureus*, 39 = *Otiorhynchus ligneus*, 40 = *Otiorhynchus ovatus*, 41 = *Otiorhynchus raucus*, 42 = *Oxypoda brachyptera*, 43 = *Oxypoda procerula*, 44 = *Ozyptila simplex*, 45 = *Paederus littoralis*, 46 = *Panagaeus bipustulatus*, 47 = *Paradromius linearis*, 48 = *Pardosa agrestis*, 49 = *Philhygra debilis*, 50 = *Philhygra palustris*, 51 = *Philorhizus melanocephalus*, 52 = *Phrurolithus minimus*, 53 = *Platyderus depressus*, 54 = *Platystethus alutaceus*, 55 = *Platystethus nitens*, 56 = *Poecilus cupreus*, 57 = *Romualdius angustisetulus*, 58 = *Rugilus orbiculatus*, 59 = *Scybalicus oblongiusculus*, 60 = *Sibianor aurocinctus*, 61 = *Sunius propinquus*, 62 = *Tasgius ater*, 63 = *Tasgius globulifer*, 64 = *Tasgius winkleri*, 65 = *Tegenaria agrestis*, 66 = *Trachyzelotes pedestris*, 67 = *Troxochrus scabriculus*, 68 = *Walckenaeria capito*, 69 = *Xantholinus elegans*, 70 = *Xysticus kochi*, 71 = *Zelotes apricorum*, 72 = *Zelotes latreillei*, 73 = *Zodarion italicum*.

cupreus (Gnaphosidae). Nine species were only recorded at Thanet mounds. Beetles were *Otiorhynchus ovatus* (Curculionidae), *Calathus ambiguus* and *Bembidion femoratum* (Carabidae). There were also two species of *Philhygra*, *Oxypoda brachyptera* and *Tasigus ater* (Staphylinidae). One spider only recorded at Thanet mounds was *Troxochrus scabriculus* (Linyphiidae).

Nine species were only recorded at RBF mounds. Beetles were two species of *Carpelimus* (Staphylinidae), *Hypera fuscocinerea* and *Romualdius angustisetulus* (Curculionidae), *Microlestes minutulus* (Carabidae), *Helophorus rufipes* (Helophoridae) and *Omphalapion hookerorum* (Apionidae). Two spiders were *Walckenaeria capito* and *Centromerus capucinus* (Linyphiidae). A slightly larger group of 13 species occurred at both RBF and Thanet mounds, but not grassland controls. These were two species of *Otiorhynchus* (Curculionidae), *Curtonotus convexiusculus* and *Platyderus depressus* (Carabidae), *Melanophthalma suturalis* (Latridiidae) and *Anthicus antherinus* (Anthicidae). There were also *Anotylus inustus*, *Astenus lyonessius*, *Lobrathium multipunctum* and *Paederus littoralis* (Staphylinidae). Three spiders were *Crustulina guttata* (Theridiidae), *Phrurolithus minimus* (Corinnidae) and *Tegenaria agrestis* (Agelenidae).

Eight stenotopic species with current UK conservation status were recorded on the mounds, of which four were not also recorded at controls. *Calathus ambiguus* was only recorded singly on a Thanet mound and may have been a vagrant. *Hypera fuscocinerea* was trapped in low numbers, with two from one RBF mound. *Centromerus capucinus* was also recorded singly on an RBF mound, and probably deserves UK conservation status. *Phrurolithus minimus* was slightly more frequent, with four on RBF mounds and three on Thanet mounds. Four species were recorded in similar numbers on mounds and controls: *Brachinus crepitans* (controls 17, RBF 38, Thanet 9); *Ophonus azureus* (controls 4, RBF 5, Thanet 2); *Sibianor aurocinctus* (controls 1, Thanet 1) and *Panagaeus bipustulatus* (controls 9, RBF 1, Thanet 2). *Harpalus anxius* was recorded once at a grassland plot. The greatest shift in abundance was for *Scybalicus oblongiusculus*. One individual was recorded in grassland plots in 2015, rising to three in 2016. There were two individuals on RBF mounds in 2015, rising to 18 in 2016, and two individuals on Thanet mounds in 2015, rising to 15 in 2016.

5.3.6 Aculeate Hymenoptera - summary

There were 53 aculeate species recorded as females either excavating or provisioning nests, or investigating suitable breeding habitat. No ground-nesting species were recorded in the grassland plots, due to the complete sward coverage and dense litter layer. A sample-based rarefaction curve estimated 62 (SE ± 11) species likely to be recorded in timed observations of the sampled habitats in 2015 (Figure 5.9). Forty-five species captured represent 72.6% of this estimated fauna. There were an estimated 43 (SE ± 7) species in the sampled habitats in 2016 (Figure 5.10). Thirty-three species captured represent 78.6% of this estimated fauna. Aculeates differed in abundance ($\chi^2_{[2, N=18]}$ = 80.48, $P < 0.0001$) between treatments in 2015, but did not differ in species richness ($\chi^2_{[2, N=18]}$ = 3.52, $P = 0.0597$) (Table 5.3). Abundance was significantly higher at Thanet than RBF mounds ($P_{adj} < 0.0001$), and richness was also higher at Thanet mounds ($P_{adj} = 0.0606$) but the difference was not significant. Aculeates differed in abundance ($\chi^2_{[2, N=18]}$ = 16.03, $P < 0.0001$) and species richness ($\chi^2_{[2, N=18]}$ = 8.09, $P = 0.0045$) between treatments in 2016. Abundance was significantly higher at Thanet mounds than RBF mounds ($P_{adj} < 0.0001$) and richness was also significantly higher at Thanet mounds ($P_{adj} = 0.005$).

Comparing abundance of aculeate Hymenoptera between Thanet and RBF mounds for each visit, permutation tests revealed significant differences in May 2015 ($P = 0.0332$), June 2015 ($P = 0.0091$), August 2015 ($P = 0.0090$) and May 2016 ($P = 0.0207$) (Table 5.4). In each case abundance was greater at Thanet mounds. In June 2016 ($P = 0.8551$) and August 2016 ($P = 0.4227$) differences in aculeate abundance between mound types were not significant. In both these visits, zero counts were recorded on seven Thanet mounds and eight RBF mounds. Substrate and site explained 20% of variation in aculeate occurrence (F=2.46, $P = 0.001$). As conditional effects, site explained the largest proportion of variation at 15% (F=2.54, $P = 0.001$), substantially higher than the 4% explained by substrate (F=2.28, $P = 0.001$). The shared variation between substrate and site was negative at -2% as the variables are complementary in explanatory power and explain the response variable better than the sum of the individual effects of these variables (Jongman, 1995).

Table 5.4 Median abundance (minimum – maximum) for aculeate Hymenoptera in aggregate mounds experiment at Hadleigh Park, Pitsea Landfill, and Rainham Marshes, in 2015 and 2016. Treatments were 5-tonne mounds of recycled bulk fill (n=9) or Thanet sand (n=9), and unmanaged grassland (n=9). This was based on 30 minutes hand netting per plot during three visits each year, in May, June, and August. Female aculeates excavating or provisioning nest holes, or quartering low over the plot surface, were captured and identified to species. Male aculeates do not form part of the analysis. Treatments were compared with the Wilcoxon matched pairs test (V) for each pair of mounds within the nine experimental blocks. There was zero data from unmanaged grassland plots.

Year	Period	RBF	Thanet	$V_{[N=9]}$	*P*-value
2015	Mid May	1 (0–4)	3 (1–9)	3	0.0332
	Late June	1 (0–5)	10 (2–21)	0	0.0091
	Early August	2 (1–5)	7 (4–19)	0	0.0090
2016	Mid May	4 (0–9)	10 (5–17)	2.5	0.0207
	Late June	0 (0–14)	0 (0–15)	4	0.8551
	Early August	0 (0–3)	0 (0–13)	1	0.4227

5.3.7 Aculeate Hymenoptera - treatments

Totals for aculeates captured at the aggregate mounds were 201 in 2015 and 146 in 2016 (Appendix 9.5, Table A9.5.3, page 334). The number of individuals decreased at the RBF mounds from 50 in 2015, to 47 in 2016. From 20 species captured in 2015, the most abundant were *Lasioglossum puncticolle* (16%), *Lasioglossum villosulum* (12%) and *Lasioglossum minutissimum* (10%). There were 16 species represented by three of fewer individuals, forming 80% of the catch. Two species were cleptoparasites (*Nomada flavoguttata* and *Sphecodes geoffrellus*). From 18 species captured in 2016, the most abundant were *Lasioglossum morio* (11%), *Lasioglossum minutissimum* (9%) and *Lasioglossum pauxillum* (9%). There were 14 species represented by three or fewer individuals, forming 78% of the catch. Three species were cleptoparasites (*Nomada flavoguttata*, *Nomada marshamella* and *Nomada goodeniana*). *Mimumesa dahlbomi* was only recorded as a male, although females are known to nest in sandy ground.

The number of individuals decreased at the Thanet mounds from 151 in 2015, to 99 in 2016. From 35 species captured in 2015, the most abundant were *Oxybelus uniglumis* (24%), *Lasioglossum minutissimum* (19%), *Nomada flavoguttata* and *Entomognathus brevis* (both 5%). There were 23 species represented by three or fewer individuals, forming 66% of the catch. Eight species were cleptoparasites (*Nomada flavoguttata*, *Nomada flavopicta*, *Nomada goodeniana*, *Sphecodes ephippius*, *Sphecodes puncticeps*, *Sphecodes gibbus*, *Sphecodes longulus* and *Sphecodes monilicornis*). From 29 species captured in 2016, the most abundant were *Lasioglossum minutissimum* (22%), *Oxybelus uniglumis* (11%) and *Andrena minutula* (8%). There were 21 species represented by three or fewer individuals, forming 72% of the catch. Seven species were cleptoparasites (*Nomada flavoguttata*, *Nomada flava*, *Nomada flavopicta*, *Nomada marshamella*, *Sphecodes geoffrellus*, *Sphecodes ephippius* and *Sphecodes puncticeps*). Seven species were only recorded as males, although females are known to nest in sandy ground (*Andrena bicolor*, *Dasypoda hirtipes*, *Sphecodes crassus*, *Crossocerus quadrimaculatus*, *Diodontus insidiosus* and *Lindenius albilabris*).

NMDS ordination based on presence/absence of aculeates (stress value 0.1417) showed assemblages at Hadleigh Park well separated on the first axis,

particularly for RBF mounds (Figure 5.13). Species associated with Thanet and RBF mounds were also separated at this site. The second axis partially separated Thanet mounds from RBF mounds, although there was considerable overlap with variability particularly from RBF mounds at Rainham. One mound here was similar to the Thanet mounds, another was more similar to RBF mounds at Pitsea, and the third was in an intermediate position between the Thanet mounds at Rainham and Hadleigh. There was slightly greater similarity in the fauna colonising Thanet mounds at Pitsea and Rainham, than between the RBF and Thanet mounds at Rainham only. NMDS based on abundance of aculeates (stress value 0.1261) was similar for the RBF mounds, although Thanet mounds were more widely separated (Figure 5.14). Thanet mounds at Pitsea and RBF mounds at Rainham partially overlapped, and were separated from other mounds at both sites.

There were 47 species at Thanet mounds and 28 species at RBF mounds, with 21 species in common to both materials. Solitary bees in common at both mounds were four species of *Andrena* (Andrenidae), Ten species of *Lasioglossum* (Halictidae) and three species of cleptoparasitic *Nomada* (Apidae). Solitary wasps in common at both mounds were *Cerceris rybyensis*, *Oxybelus uniglumis* and *Trypoxylon medium* (Crabronidae). Twenty-seven species were only recorded at Thanet mounds. Solitary bees were *Lasioglossum leucozonium*, *Halictus rubicundus* and six species of cleptoparasitic *Sphecodes* (Halicitdae), *Melitta leporina* (Melittidae), five species of *Andrena* (Andrenidae) and two species of cleptoparasitic *Nomada* (Apidae). Solitary wasps were two species of *Cerceris*, *Crossocerus wesmaeli*, *Entomognathus brevis*, *Gorytes quadrifasciatus*, *Mimumesa unicolor* and *Philanthus triangulum* (Crabronidae), *Arachnospila anceps*, *Priocnemis pusilla* and *Episyron rufipes* (Pompilidae) and *Ammophila sabulosa* (Sphecidae). Seven species were only recorded at RBF mounds. Solitary bees were *Sphecodes geoffrellus* (Halictidae), *Anthophora bimaculata* (Apidae) and two species of *Hylaeus* (Colletidae). Solitary wasps were *Anoplius concinnus* and *Auplopus carbonarius* (Pompilidae).

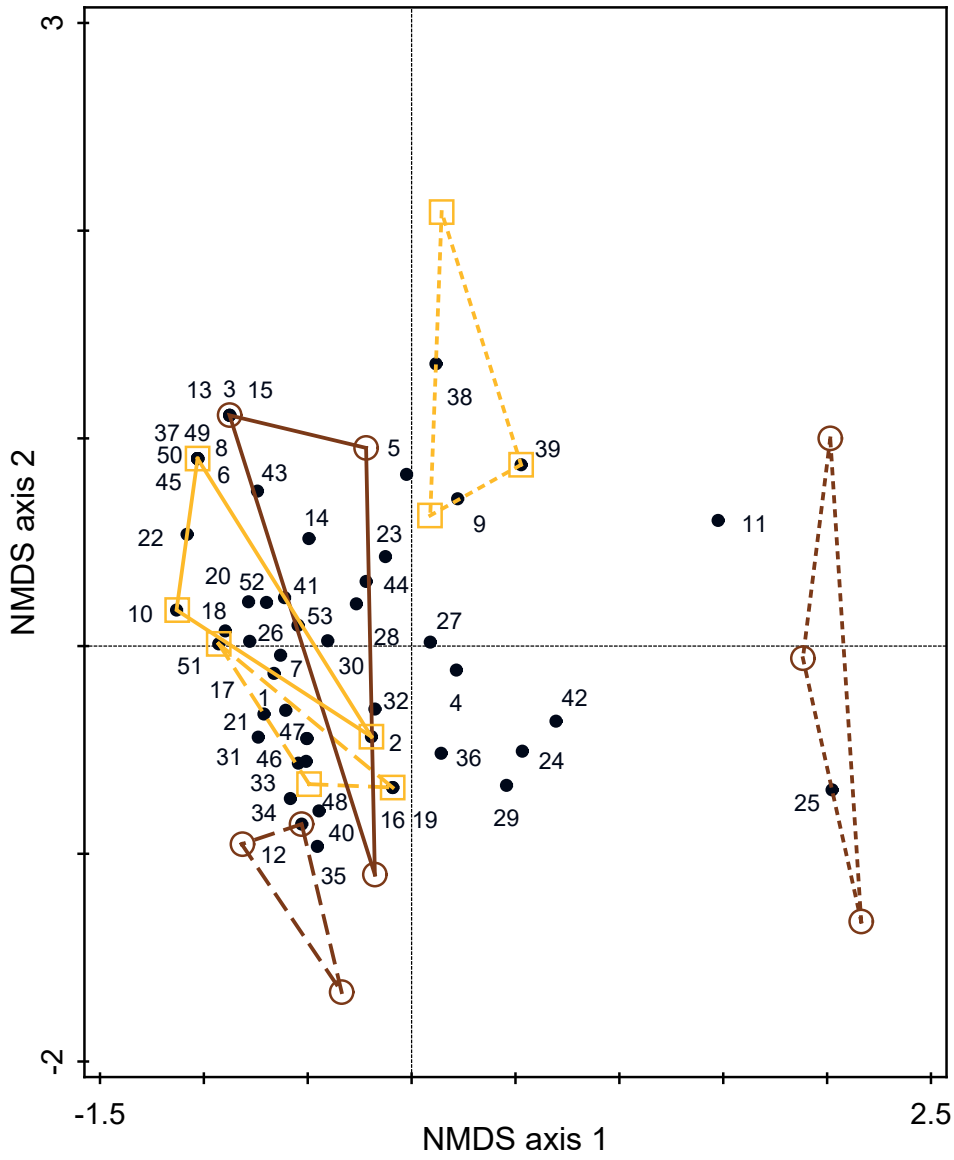

Figure 5.13 Biplot scores for aculeate Hymenoptera species (●) and treatment plots, calculated from Non-metric Multidimensional Scaling of presence-absence data at aggregate mounds in south Essex, 2015-2016 (stress value = 0.1417, 3 dimensions). Ordination was based on a matrix of Jaccard dissimilarities, with a maximum of 100 random starts to search for the lowest stress solution. Samples were pooled from both years, based on 30 minutes hand netting per plot (n=27), during three visits in May, June, and August. Treatments were 5-tonne mounds of recycled bulk fill or Thanet sand, and unmanaged grassland. There was zero data from unmanaged grassland plots, and these were not included in the ordation. Treatments are coded recycled bulk fill (◯), Thanet sand (□). Sites are coded Hadleigh Park (- - - -), Pitsea Landfill (— —), Rainham Marshes (——). Numerical references for species names: 1 = *Ammophila sabulosa*, 2 = *Andrena chrysosceles*, 3 = *Andrena dorsata*, 4 = *Andrena flavipes*, 5 = *Andrena haemorrhoa*, 6 = *Andrena labialis*, 7 = *Andrena minutula*, 8 = *Andrena nigroaenea*, 9 = *Andrena nitida*, 10 = *Andrena praecox*, 11 = *Andrena scotica*, 12 = *Anoplius concinnus*, 13 = *Anthophora bimaculata*, 14 = *Arachnospila anceps*, 15 = *Auplopus carbonarius*, 16 = *Cerceris arenaria*, 17 = *Cerceris quinquefasciata*, 18 = *Cerceris rybyensis*, 19 = *Crossocerus wesmaeli*, 20 = *Entomognathus brevis*, 21 = *Episyron rufipes*, 22 = *Gorytes quadrifasciatus*, 23 = *Halictus rubicundus*, 24 = *Hylaeus communis*, 25 = *Hylaeus hyalinatus*, 26 = *Lasioglossum albipes*, 27 = *Lasioglossum calceatum*, 28 = *Lasioglossum leucozonium*, 29 = *Lasioglossum malachurum*, 30 = *Lasioglossum minutissimum*, 31 = *Lasioglossum morio*, 32 = *Lasioglossum parvulum*, 33 = *Lasioglossum pauperatum*, 34 = *Lasioglossum pauxillum*, 35 = *Lasioglossum puncticolle*, 36 = *Lasioglossum villosulum*, 37 = *Melitta leporina*, 38 = *Mimumesa unicolor*, 39 = *Nomada flava*, 40 = *Nomada flavoguttata*, 41 = *Nomada flavopicta*, 42 = *Nomada goodeniana*, 43 = *Nomada marshamella*, 44 = *Oxybelus uniglumis*, 45 = *Philanthus triangulum*, 46 = *Priocnemis pusilla*, 47 = *Sphecodes ephippius*, 48 = *Sphecodes geoffrellus*, 49 = *Sphecodes gibbus*, 50 = *Sphecodes longulus*, 51 = *Sphecodes monilicornis*, 52 = *Sphecodes puncticeps*, 53 = *Trypoxylon medium*.

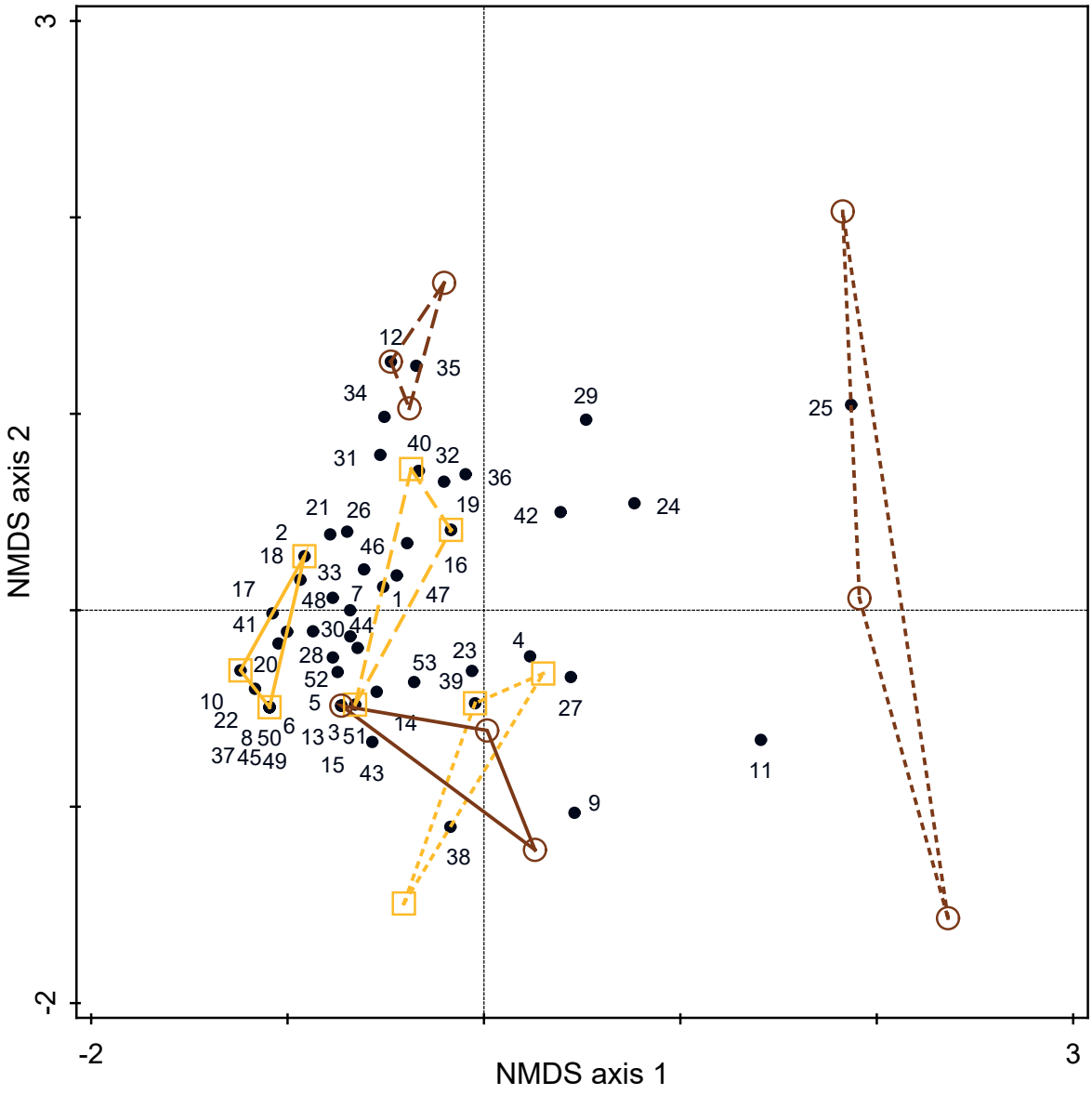

Figure 5.14 Biplot scores for aculeate Hymenoptera species (●) and treatment plots, calculated from Non-metric Multidimensional Scaling of abundance data at aggregate mounds in south Essex, 2015-2016 (stress value = 0.1261, 3 dimensions). Ordination was based on a matrix of Bray-Curtis dissimilarities, with a maximum of 100 random starts to search for the lowest stress solution. Samples were pooled from both years, based on 30 minutes hand netting per plot, during three visits in May, June, and August. Treatments were 5-tonne mounds of recycled bulk fill (n=9) or Thanet sand (n=9), and unmanaged grassland (n=9). There was zero data from unmanaged grassland plots, and these were not included in the ordation. Treatments are coded recycled bulk fill (◯), Thanet sand (□). Sites are coded Hadleigh Park (- - - -), Pitsea Landfill (— —), Rainham Marshes (———). Numerical references for species names: 1 = *Ammophila sabulosa*, 2 = *Andrena chrysosceles*, 3 = *Andrena dorsata*, 4 = *Andrena flavipes*, 5 = *Andrena haemorrhoa*, 6 = *Andrena labialis*, 7 = *Andrena minutula*, 8 = *Andrena nigroaenea*, 9 = *Andrena nitida*, 10 = *Andrena praecox*, 11 = *Andrena scotica*, 12 = *Anoplius concinnus*, 13 = *Anthophora bimaculata*, 14 = *Arachnospila anceps*, 15 = *Auplopus carbonarius*, 16 = *Cerceris arenaria*, 17 = *Cerceris quinquefasciata*, 18 = *Cerceris rybyensis*, 19 = *Crossocerus wesmaeli*, 20 = *Entomognathus brevis*, 21 = *Episyron rufipes*, 22 = *Gorytes quadrifasciatus*, 23 = *Halictus rubicundus*, 24 = *Hylaeus communis*, 25 = *Hylaeus hyalinatus*, 26 = *Lasioglossum albipes*, 27 = *Lasioglossum calceatum*, 28 = *Lasioglossum leucozonium*, 29 = *Lasioglossum malachurum*, 30 = *Lasioglossum minutissimum*, 31 = *Lasioglossum morio*, 32 = *Lasioglossum parvulum*, 33 = *Lasioglossum pauperatum*, 34 = *Lasioglossum pauxillum*, 35 = *Lasioglossum puncticolle*, 36 = *Lasioglossum villosulum*, 37 = *Melitta leporina*, 38 = *Mimumesa unicolor*, 39 = *Nomada flava*, 40 = *Nomada flavoguttata*, 41 = *Nomada flavopicta*, 42 = *Nomada goodeniana*, 43 = *Nomada marshamella*, 44 = *Oxybelus uniglumis*, 45 = *Philanthus triangulum*, 46 = *Priocnemis pusilla*, 47 = *Sphecodes ephippius*, 48 = *Sphecodes geoffrellus*, 49 = *Sphecodes gibbus*, 50 = *Sphecodes longulus*, 51 = *Sphecodes monilicornis*, 52 = *Sphecodes puncticeps*, 53 = *Trypoxylon medium*.

5.3.8 Functional groups of beetles and spiders

Phytophagous beetles differed significantly in abundance ($\chi^2_{[2, N=27]} = 9.35$, $P = 0.0093$) and richness ($\chi^2_{[2, N=27]} = 7.07$, $P = 0.0291$) between treatments in 2015 (Table 5.5). Abundance of phytophagous beetles was significantly higher at RBF mounds than Thanet mounds ($P_{adj} = 0.0216$). Species richness of phytophagous beetles was significantly higher at RBF mounds than control plots ($P_{adj} = 0.0461$). Phytophagous beetles also differed significantly in abundance ($\chi^2_{[2, N=27]} = 25.88$, $P < 0.0001$) and richness ($\chi^2_{[2, N=27]} = 7.07$, $P = 23.21$) between treatments in 2016. Abundance of phytophagous beetles was significantly higher at RBF mounds than control plots ($P_{adj} < 0.0001$) or Thanet mounds ($P_{adj} = 0.0103$). Species richness of phytophagous beetles was significantly greater at RBF mounds ($P_{adj} < 0.0001$) and Thanet mounds ($P_{adj} = 0.0011$) compared with control plots.

Predatory species differed significantly in abundance ($\chi^2_{[2, N=27]} = 120.15$, $P < 0.0001$) and richness ($\chi^2_{[2, N=27]} = 12.15$, $P = 0.0023$) between treatments in 2015. Abundance of predatory species was significantly higher at control plots than RBF or Thanet mounds ($P_{adj} < 0.0001$ in both cases), and at RBF than Thanet mounds ($P_{adj} = 0.0008$). Species richness of predators was significantly higher at control plots than Thanet mounds ($P_{adj} = 0.0015$). Predators also differed significantly in abundance ($\chi^2_{[2, N=27]} = 54.23$, $P < 0.0001$) and richness ($\chi^2_{[2, N=27]} = 2.88$, $P = 0.0201$) between treatments in 2016. Abundance of predators was significantly higher at control plots than RBF or Thanet mounds ($P_{adj} < 0.0001$ in both cases). Species richness of predators was significantly higher at control plots than Thanet mounds ($P_{adj} = 0.0461$).

Zoophagous beetles differed significantly in abundance ($\chi^2_{[2, N=27]} = 19.27$, $P < 0.0001$) between treatments in 2015, but not in species richness ($\chi^2_{[2, N=27]} = 4.81$, $P = 0.2593$). Abundance of zoophagous beetles was significantly higher at control plots than Thanet mounds ($P_{adj} = 0.0001$). Zoophagous beetles differed significantly in abundance ($\chi^2_{[2, N=27]} = 9.11$, $P < 0.0011$) between treatments in 2016, but not in species richness ($\chi^2_{[2, N=27]} = 0.93$, $P = 0.1466$). Abundance of zoophagous beetles was significantly higher at control plots ($P_{adj} < 0.0001$) and RBF mounds ($P_{adj} = 0.0216$) than Thanet mounds. Detritivorous beetles did not differ significantly in abundance ($\chi^2_{[2, N=27]} = 2.48$, $P = 0.2898$) or richness ($\chi^2_{[2,$

Table 5.5 Mean ± standard error for species richness and abundance of beetle and spider functional groups in aggregate mounds experiment. Samples were pooled from three pitfall traps within each monitoring plot (n=27), open for 14 days in May, July, and September, 2015-2016. Generalized linear mixed models (χ^2, F) compared predatory (beetle and spider), zoophagous, phytophagous, and detritivorous groups (beetles only) at 5-tonne piles of recycled bulk fill or Thanet sand, and unmanaged grassland controls. Post-hoc tests show Tukey pairwise comparisons after Holm-Šidák adjustment, treatments with shared superscript a-c (ranked highest to lowest) did not differ significantly ($P > 0.05$). Moran's I was calculated to test for spatial correlation in GLMM residuals, based on a matrix of inverse Euclidean distance weights (significance level $P < 0.05$).

Functional group	Year	Variable	Control	RBF	Thanet	$GLMM_{[2, N=27]}$	Moran's I_{GLMM}
Predatory	2015	Abundance	237.7 ± 22.4 ^{a}	111.7 ± 10.4 ^{b}	74.6 ± 9.9 ^{b}	$\chi^2 = 120.15$	0.5281
		Species richness	40.4 ± 3.1 ^{a}	35.3 ± 2.1 ^{ab}	30.7 ± 1.7 ^{b}	$F = 12.15$	0.7601
	2016	Abundance	294.2 ± 33.1 ^{a}	130.6 ± 10.0 ^{b}	140.3 ± 16.3 ^{b}	$\chi^2 = 54.23$	0.5679
		Species richness	46.1 ± 3.1 ^{a}	38.8 ± 2.5 ^{ab}	38.7 ± 1.8 ^{b}	$F = 7.81$	0.4927
Zoophagous	2015	Abundance	24.0 ± 3.8 ^{a}	16.8 ± 2.3 ^{b}	10.1 ± 1.6 ^{c}	$\chi^2 = 19.27$	0.1209
		Species richness	8.3 ± 0.8	6.2 ± 0.4	5.8 ± 0.5	$F = 4.8121$	0.2593
	2016	Abundance	54.1 ± 7.6 ^{a}	48.0 ± 11.4 ^{a}	32.0 ± 5.8 ^{b}	$\chi^2 = 9.11$	0.3940
		Species richness	7.9 ± 0.9	7.9 ± 0.5	8.3 ± 1.1	$F = 0.93$	0.5308
Phytophagous	2015	Abundance	43.6 ± 12.0 ^{ab}	64.8 ± 25.2 ^{a}	24.7 ± 5.1 ^{b}	$\chi^2 = 9.35$	0.2124
		Species richness	11.4 ± 0.8 ^{b}	15.7 ± 1.3 ^{a}	12.0 ± 1.2 ^{ab}	$F = 7.07$	0.6061
	2016	Abundance	52.7 ± 11.3 ^{b}	156.6 ± 20.5 ^{a}	82.8 ± 12.2 ^{b}	$\chi^2 = 25.88$	0.9859
		Species richness	13.8 ± 1.7 ^{b}	27.2 ± 2.4 ^{a}	23.3 ± 2.4 ^{a}	$\chi^2 = 23.21$	0.0564
Detritivorous	2015	Abundance	14.6 ± 3.2	11.4 ± 2.1	8.8 ± 1.9	$\chi^2 = 2.48$	0.3172
		Species richness	11.44 ± 1.5	16.2 ± 3.1	12.8 ± 2.3	$F = 2.22$	0.7767
	2016	Abundance	6.3 ± 1.2	5.9 ± 0.7	4.9 ± 1.1	$\chi^2 = 1.76$	0.6219
		Species richness	5.3 ± 0.6	7.3 ± 0.9	7.0 ± 1.2	$F = 3.26$	0.4362

$_{N=27]}$ = 1.76, P = 0.4151) between treatments in 2015. There was also no significant difference in abundance ($\chi^2_{[2, N=27]}$ = 2.22, P = 0.3325) or richness ($\chi^2_{[2, N=27]}$ = 3.26, P = 0.1963) of detritivorous beetles between treatments in 2016.

5.3.9 Association of environmental variables and functional groups of beetles and spiders

For the RDA ordination in 2015 (Figure 5.15), axis one explained 43.58% of the variation in functional groups. Axis two explained 11.19% of the variation, and 54.77% in combination with axis one. From the seven environmental variables, three contributed significantly to explain variation in functional group occurrence: percentage of bare ground (negatively correlated with axes one and two, F-ratio = 8.9, P_{adj} = 0.014), graminoid species richness (positively correlated with axis 1, F-ratio = 6.9, P_{adj} = 0.024) and forb species richness (negatively correlated with axis 2, F-ratio = 5.9, P_{adj} = 0.014). Other environmental variables either correlated with these three, or did not describe a significant proportion of the variation in the ordination.

Axis one separated Thanet mounds from grassland controls, with RBF mounds between and partially overlapping these. All functional groups were positively associated with axis one, particularly predatory beetles, predatory spiders, zoophagous beetles, and detritivores. On axis two, greater overall plant species richness and richness of annual plants were negatively associated with axis two. Greater graminoid species richness was positively associated with axis two. Axis two also separated low litter values at RBF mounds from grassland controls with high litter values. There was overlap with intermediate litter values recorded at samples from all treatments. Phytophagous beetles were negatively associated with axis two.

For the RDA ordination in 2016 (Figure 5.16), axis one explained 29.85% of variation in functional groups. Axis two explained 9.64% of the variation, and 39.49% in combination with axis one. From the seven environmental variables, one contributed significantly to explain variation in functional group occurrence: litter depth (positively correlated with axis one, F-ratio = 7.4, P_{adj} = 0.014). Other environmental variables either correlated with this, or did not describe a significant proportion of the variation in the ordination. Axis one separated most

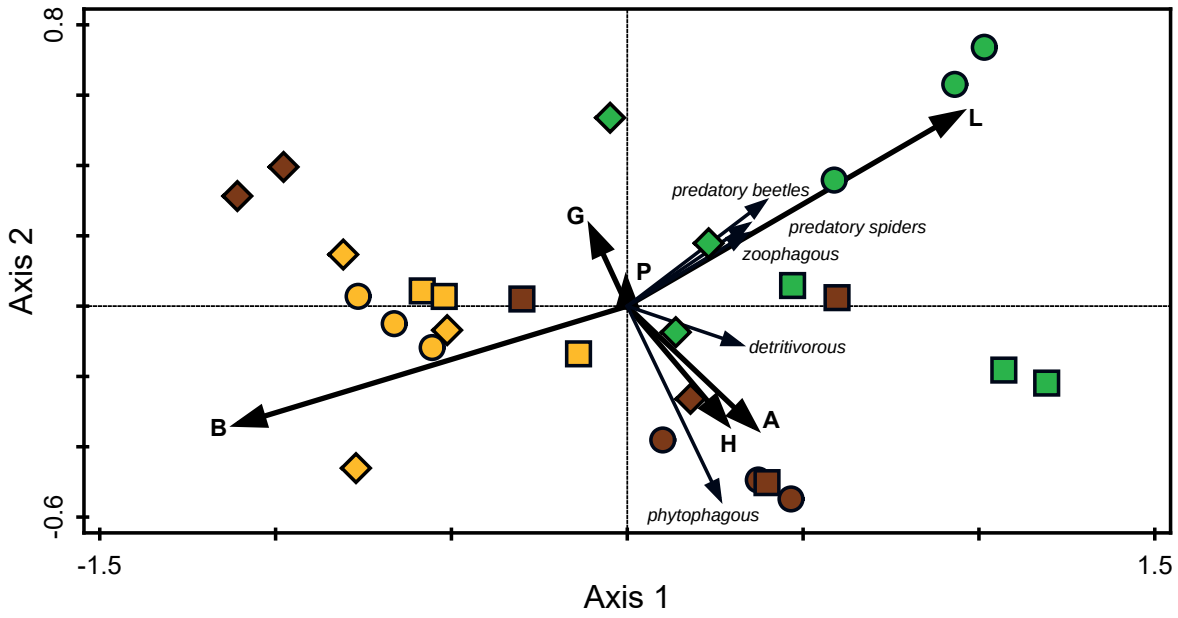

Figure 5.15 Biplot scores for invertebrate functional groups (→), environmental variables (→), and treatment plots, calculated with Redundancy Analysis (RDA) of beetle and spider assemblages from the aggregate mound creation experiment in 2015. Samples were pooled from two pitfall traps per monitoring plot (n=27), open for 14 days in May, July, and September. Treatments were 5-tonne mounds of recycled bulk fill or Thanet sand, and unmanaged grassland controls. Treatments are coded recycled bulk fill (◆ ■ ●), Thanet sand (◆ ■ ●), unmanaged grassland (◆ ■ ●). Sites are coded Hadleigh Park (■ ■ ■), Pitsea Landfill (● ● ●), Rainham Marshes (◆ ◆ ◆). Environmental variables are coded B, bare ground; H, herbaceous plant species richness; A, annual plant species richness; P, perennial plant species richess; G, graminoid species richess; L, litter depth. RDA scaling was focused on inter-species correlations and centred by species. Treatments and sites were included in the final triplot but were not used to influence the analysis. Relationships between environmental variables and distribution of functional groups were tested by forward selection (Monte Carlo test, 500 permutations). Permutations were made within but not between whole plots that were defined by sites. Significant variables were percentage of bare ground (P_{adj} = 0.014), graminoid species richness (P_{adj} = 0.024) and herbaceous plant species richness (P_{adj} = 0.014). Other environmental variables either correlated with these three, or did not describe a significant proportion of the variation in the ordination.

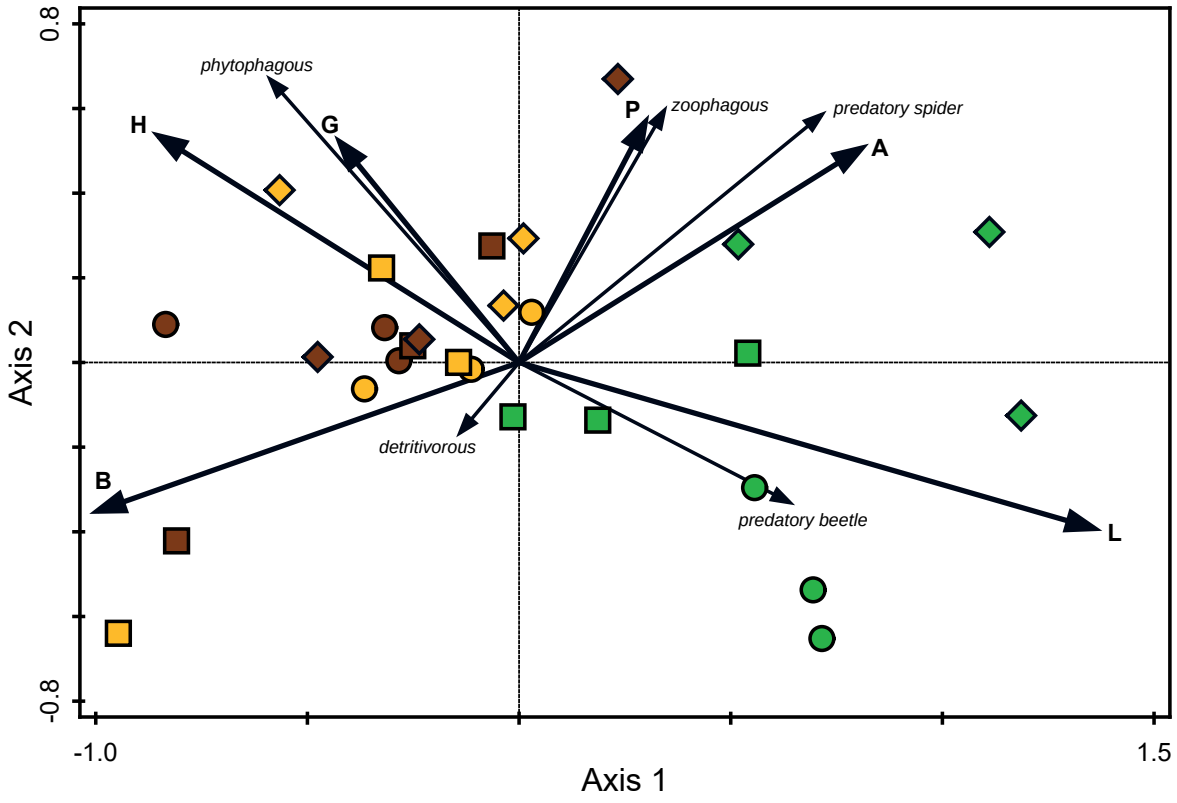

Figure 5.16 Biplot scores for invertebrate functional groups (→), environmental variables (→), and treatment plots, calculated with Redundancy Analysis (RDA) of beetle and spider assemblages from the aggregate mounds experiment in 2016. Samples were pooled from two pitfall traps per monitoring plot (n=27), open for 14 days in May, July, and September. Treatments were 5-tonne mounds of recycled bulk fill or Thanet sand, and unmanaged grassland controls. Treatments are coded recycled bulk fill (◆ ■ ●), Thanet sand (◇ □ ○), unmanaged grassland (◆ ■ ●). Sites are coded Hadleigh Park (■ □ ■), Pitsea Landfill (● ○ ●), Rainham Marshes (◆ ◇ ◆). Environmental variables are coded B, bare ground; H, herbaceous plant species richness; A, annual plant species richness; P, perennial plant species richess; G, graminoid species richess; L, litter depth. RDA scaling was focused on inter-species correlations and centred by species. Treatments and sites were included in the final triplot but were not used to influence the analysis. Relationships between environmental variables and distribution of functional groups were tested by forward selection (Monte Carlo test, 500 permutations). Permutations were made within but not between whole plots that were defined by sites. The significant variable was litter depth (P_{adj} = 0.014). Other environmental variables either correlated with this, or did not describe a significant proportion of the variation in the ordination.

of the aggregate mounds from the grassland controls, with a tendency for RBF mounds to be more strongly negatively associated, but overlapping with the Thanet mounds.

Measures of plant species richness were positively associated with axis 2, including most aggregate mounds. The grassland plots were variable with respect to axis two. Species-poor samples from Hadleigh were negatively associated with axis two, particularly compared with two samples from Rainham and one sample from Pitsea. Predatory beetles and spiders were particularly positively associated with axis one. Zoophagous beetles were positively associated with axis two and axis one. Detritivores were negatively associated with axis one and axis two. Phytophagous beetles were negatively associated with axis one and positively associated with axis two.

5.4 Discussion

5.4.1 Early successional habitat creation

This study focused on the effects of substrate addition in areas of unmanaged grassland. In the first year, aggregate mounds attracted aculeate Hymenoptera, with the greatest abundance and richness of species associated with Thanet sand. Although abundance and richness were lower at RBF mounds, there were distinct groups of species associated with each substrate. Stenotopic beetles and spiders were less abundant on Thanet mounds in the first year, and species richness was not altered compared with the existing grassland. Abundance of predatory and zoophagous Coleoptera was greatest in the grassland. Abundance of phytophagous Coleoptera was reduced on Thanet mounds, and abundance of detritivores did not differ between treatments.

In the second year, there were fewer aculeate Hymenoptera overall and most were recorded in May. Stenotopic beetles and spiders were more abundant on RBF mounds, although species richness was similar compared with the existing grassland. Abundance of predatory invertebrates was greatest in the grassland plots, although species richness did not differ between treatments. Abundance of phytophagous Coleoptera was greatest on RBF mounds, and species richness was higher on both mound substrates compared with the existing grassland. Abundance and species richness of detritivores did not differ between treatments.

5.4.2 Development of created habitat

In this study, vegetative growth of perennial plants was at least twice as frequent on mounds as colonisation by annual seedlings, with *Cirsium arvense* and *Elytrigia repens* particularly prominent. Grasses were most frequent around the perimeter of the mound, while *Convolvulus arvensis* and *Cirsium arvense* were the most successful forbs colonising the middle slopes and mound tops. On 75 cm diameter, 15 cm high earth mounds created in prairie grasslands, species composition closely resembled surrounding vegetation after a similar pattern of colonisation (Umbanhowar, 1992). A year after mound creation there were 512 stems, compared with 42 seedlings, which was partly attributed to the creation of mounds from subsoil where the seedbank might have been depleted (Umbanhowar, 1992). Growth form might also be an important factor explaining the concentration of grasses at the perimeter. Forbs grow from apical meristems, whereas grasses grow from basal meristems and would have to push the entire plant through the soil (Umbanhowar, 1992).

The other major difference between the vegetation of mounds and controls was the establishment of annuals in the first and second years on RBF mounds, and the second year on Thanet mounds. The Thanet sand was a freshly quarried Thanetian deposit (Ellison *et al.*, 2004) but the RBF was not sterilised and it is likely that seeds were introduced to the site (although seed banks were not tested). By the second year, the large gaps available on both mounds were colonised by annual and perennial forbs, resulting in greater species richness than the grassland. Colonisation by early successional ruderal plants was observed in chalk grassland restoration on ex-arable land, without addition of sowing treatments (Mortimer *et al.*, 2002). Ruderal forbs also established in a mixture of sand, silt and composted material, compared with seeded prairie species on mounds of quarry sand (Biederman and Whisenant, 2011). Overall in the present study, development of vegetation affected the amount of bare ground on both mound types, which had been extensively bare in the first year. By the second year only small patches of open bare ground remained.

5.4.3 Response of aculeates

The unmanaged grassland provided limited opportunities for aculeate Hymenoptera nesting in bare substrate. Increased activity associated with

freshly deposited aggregate was in line with expectations. This provides further evidence of the critical role of nesting resources for burrowing aculeates, which has been identified as a limitation in the effectiveness of current agri-environment schemes (Nieto *et al.*, 2014). Thirty seven bee species colonised in two years after experimental substrate amendment in France, with 40% only recorded as single specimens (Fortel *et al.*, 2016). There were 12 species in common with the present study, in the genera *Andrena* (three species), *Lasioglossum* (six species), *Nomada* (one species) and *Sphecodes* (two species). Rapid colonisation by aculeates was also observed a year after restoration of alluvial dunes in Germany (Exeler *et al.*, 2009). In the present study there were no areas of exposed sand located within sight of the experimental plots to act as an obvious source of colonising aculeate Hymenoptera, although in a UK context land on the northern side of the Thames Estuary supports a very diverse aculeate assemblage. 96% of the Essex aculeate fauna and 74% of the national fauna have been recorded in the area (Harvey, 2000).

At Rainham there were previously disturbed sand banks 700 m west of the experimental area, which were part of a World War Two firing range. Grazing was excluded some time before the present study, which resulted in a decline in available bare ground. During 2015-2016 some sandy substrate remained at the inactive Wennington silt lagoons (2-3 km west). The nearest large area of aculeate habitat to Pitsea was 3.2 km south-east at Canvey Wick, a Site of Special Scientific Interest created by onshore disposal of marine dredgings. At Hadleigh Park, conservation management since 2015 created small sand exposures at Hadleigh Downs, 900 m north of the grazing marsh. *Lasioglossum malachurum* also nested about 100 m from the experimental area, in compacted bare soil outside a field gateway, but the extent to which smaller aculeate populations were scattered around the experimental areas is unknown. In comparison, 25 species of Sphecidae and four species of Pompilidae colonised a newly exposed sand bank within two years (Evans, 1974). 13 species were already present on an adjacent plot and most others were presumed to have colonised from beyond the immediate locality.

Different species assemblages were associated with the two mound types and although detailed autecological information is not available for all aculeate

Hymenoptera, this is consistent with most previous research. Niche specification by substrate was documented for thirty-two species of fossorial bees in the United States, with sand, silt loam and clay loam soils all occupied but no specific familial or tribal affiliations (Cane, 1992). In the Czech Republic, *Sphex funerarius* was associated with increased gravel content, *Ammophila sabulosa* and *Ammophila pubescens* with sand grains, and *Ammophila heydeni* was present across a broad range of soils (Srba and Heneberg, 2012). One experimental study found no effect of substrate on colonising bees, for sand and clay mixed with local soils (Fortel *et al.*, 2016). There was low sampling intensity in this study, which might have limited the scope to detect differences, and only flat areas of bare ground were created. Colonisation by solitary wasps was also not considered.

While different preferences were apparent for some aculeates associated with the mounds, this does not imply limits on the conditions tolerated. The short time for observations and reliance on hand netting will not have produced a comprehensive inventory of the species present. In addition, a factor can only be causal when it directly affects the structure of a nest, or female behaviour associated with nest-site selection (Potts and Willmer, 1997). *Ammophila sabulosa* was only recorded on Thanet mounds in this study. Prey theft and brood parasitism are the most common nesting strategies for this species, and parasitic miltogrammine flies larviposit into nest burrows (Field, 1989). Nest entrances are closed by foraging females, which was probably easier on Thanet mounds.

There was greater abundance and diversity of aculeates at Thanet mounds in both years. A noticeable difference between the two aggregates was that RBF developed a hard crust as it dried in the summer. Moist soils with clay or silt content form a dry surface crust in high evaporative conditions, due to suspension of fine particles under capillary action from beneath (Awadhwal and Thierstein, 1985). Some species may prefer softer substrate partly due to the time and effort needed to excavate hard soils, and resulting wear to mandibles, wings and cuticle (Alcock, 1980; Evans & Hook, 1986a). Effects of cuticle abrasion were illustrated for desert ants by a two to three-fold increase in water loss rates, for workers and foundresses exposed to soil during nest excavation (Johnson, 2000).

The increased difficulty of nest excavation was documented for *Mellinus arvensis*, which shifted from digging in softer soils, to searching for unoccupied burrows in harder soils (Ghazoul, 2001). The associated time costs were 1,735 min, compared with 130 min for digging a new burrow or searching for an abandoned nest (Ghazoul, 2001). There was also a greater risk of injury due to more aggressive encounters with females in occupied burrows (Ghazoul, 2001). The effect of substrate hardness on burrowing has been demonstrated experimentally for the sphecid wasp *Cerceris antipodes*, which preferentially initiated nests after watering the ground (McCorquodale, 1989).

Although species richness and abundance were lower at RBF mounds, some aculeates were particularly associated with this material. *Anoplius concinnus* was observed stocking a nest beneath stones with *Trochosa ruricola* (Lycosidae). This species excavates multiple nest cells beneath stones and is usually found in stony or gravelly places such as riversides (Day, 1988). *Lasioglossum malachurum* colonised the level mound tops. In arable, urban and woodland situations this species nested in hard packed, partially vegetated soils (Polidori *et al.*, 2010). There are also examples of aculeates selecting compacted substrates in other faunas. *Ammophila braunsi* was unique amongst six Afrotropical species in its use of abandoned burrows of other wasps in non-friable clay soils, particularly the eumenid *Parachilus insignis* (Weaving, 1989). *Dieunomia triangulifera* (Halictidae) was found to prefer compact soil, nesting preferentially in a Kansas dirt road compared with adjacent arable margins (Wuellner, 1999).

As extensive vegetation grew through and over the mounds in 2016, the number of aculeates attempting to nest reduced dramatically. Zero counts were obtained from all mounds with near complete vegetation cover in the last two visits, even though there were large areas of bare soil beneath the shade of taller plants. Although individual species nest in areas with variable vegetation cover, these rely on at least partially insolated bare ground (Heneberg *et al.*, 2013). This has been demonstrated experimentally by the complete absence of nest foundation in a patch of artificially shaded ground, which had previously been highly productive (Potts and Willmer, 1997).

There are several possible reasons for the importance of partly bare ground to nesting aculeate Hymenoptera. Even with short vegetation, roots may cause

direct mortality in nest cells. Tall vegetation is likely to make the location of nests more difficult and cooler temperatures in shade might not be optimal for development of eggs and larvae. In a study of 10 UK colonies of *Halictus rubicundus*, areas of maximum nest density had significantly greater soil temperature excess than those with nests absent (Potts and Willmer, 1997). A warmer nest could allow adults to reach the threshold required for flight more quickly, making more time available for nest excavation, mating and foraging (Stone, 1994). Temperature also influences brood survivorship. Larvae of *Philanthus triangulum* that grew up at 20°C had a 51% probability of surviving until emergence, compared with a survival chance of 69% at 25°C (Strohm, 2000). This was possibly linked to microbial infestation, which might limit the time a larva can feed without being killed by fungi or bacteria (Strohm, 2000).

5.4.4 Response of stenotopic beetles and spiders

NMDS ordination showed considerable similarity between species composition on the mounds and in the surrounding grassland. Overall abundance of stenotopic species on mounds increased in the second year, particularly at RBF mounds. One reason for this was the number of at least partly graminivorous carabids, with an eight-fold increase in Zabrini, and a two-fold increase in Harpalini (Talarico *et al.*, 2016). These were probably associated with annual plants producing abundant seed on the mounds (Harvey *et al.*, 2008), including *Hirschfeldia incana* and *Senecio inaequidens*. Other species might have responded to structural features in the habitat. There were 29 specimens of the woodlouse-hunting spider *Dysdera crocata* at RBF mounds, compared with three at both grassland plots and Thanet mounds. It spins a silk retreat on the underside of stones (Hopkin and Martin, 1985), which were only frequent at RBF mounds. There was a similar pattern in occurrence of *Zodarion italicum*, with 23 specimens on RBF mounds, compared with one at grassland plots, and six at Thanet mounds. This species also spins an igloo-like silk retreat and places egg-sacs on the underside of stones, brick and concrete debris (Harvey and Murphy, 1985).

Eight species with conservation status associated with early successional habitats were recorded on mounds, with five also recorded in the unmanaged grassland. This probably reflects the extreme climatic conditions in south-east Essex, which is the driest part of the UK, with frequent soil water deficit between

May and August (Jermyn, 1974). *Calathus ambiguus* was recorded as a single specimen on a Thanet mound, possibly as a vagrant. This carabid occurred in dunes (Desender, 1996) and was also identified as a stenotopic species on sandy road side verges in the Netherlands (Vermeulen, 1993). *Hypera fuscocinerea* was the only species with conservation status recorded on RBF mounds but not elsewhere. It is particularly associated with dry, open situations in the UK (Morris, 2002) and probably occurred following establishment of a larval foodplant on the mounds, *Medicago sativa* (Fabaceae). *Centromerus capucinus* was an additional spider only recorded on an RBF mound at Hadleigh. This lacks current conservation status but was the first UK record since 2004. The ecological preferences are poorly known, including sandy grassland and undercliffs (Harvey *et al.*, 2002).

Two species were captured in greater numbers in both years. *Phrurolithus minimus* occurred on RBF and Thanet mounds, and has previously been noted as an endangered spider in a survey of quarries in the Czech Republic (Tropek and Konvicka, 2008). *Scybalicus oblongiusculus* did occur in low numbers at control plots (4) as well as mounds (4) in the first year, but in the second year was clearly more frequent on both mound types (33). This species was thought to be extinct in the UK but was rediscovered in Essex in 2002 and subsequently particularly associated with sparsely vegetated dry grasslands, including at former silt lagoons and fly ash deposits (Hammond and Harvey, 2008). The mounds might provide more xeric conditions apparently favoured in the UK, or a concentrated seed resource compared with the grassland.

Although the grassland was of low botanical interest, there were invertebrates of conservation interest not found at the mounds. *Brachida exigua* occurred as a single specimen at Hadleigh. This staphylinid is poorly known in the UK, and was classified as Red Data Book status Indeterminate (Hyman and Parsons, 1994). It has been reported from chalk grassland in dense swards of *Brachypodium pinnatum* after cessation of grazing, which provided a moist litter layer at a warm, south-facing, well-drained site (Welch, 1989). There is more general conservation interest in field boundaries as a refuge for farmland arthropods (Lagerlöf and Wallin, 1993), and nest-searching queen bumblebees were strongly associated with tussock-type vegetation (Kells and Goulson, 2003). It is important that any creative conservation is based on a thorough

understanding of the invertebrate fauna at a site, and at a scale that will not disrupt existing valuable habitat. In this project, unmanaged grassland was continuous at each site over at least 0.5 ha, and aggregate was added to 0.0006 ha in total.

5.4.5 Response of beetle and spider functional groups

There is little information available about the effects of substrate amendment on invertebrates. A study of restoration at open-cast lignite mines patterned with crests and troughs found higher densities of most arthropod groups in troughs (Topp *et al.*, 2001), although in this case large areas were modified. Diversity measured by Fisher's alpha index was also greater where there was a pattern of crests and troughs 2 m high and with 2–4 m spacing, compared with levelled areas (Topp *et al.*, 2010). Vegetated earth banks as a topographic modification have been investigated in agricultural environments, to increase populations of overwintering predatory invertebrates. These were found to support carabids, chrysomelids and weevils at densities approaching established permanent field margins (Thomas *et al.*, 2001).

There was a significant increase in species richness and abundance of phytophagous beetles after substrate amendment in this study, particularly on RBF mounds. This can partly be attributed to establishment of plants. *Hirschfeldia incana* (Brassicaceae) was a potential host plant for five common species of leaf beetle (Chrysomelidae) that were abundant, and absent from control plots: *Phyllotreta atra* (257), *Phyllotreta consobrina* (100), *Phyllotreta diademata* (87), *Phyllotreta nigripes* (75) and *Phyllotreta undulata* (103). *Senecio inaequdiens* (Asteraceae) was a potential host plant for *Longitarsus dorsalis* (241 specimens from mounds compared with two specimens in control plots). Similar effects were found in studies of invertebrate colonisation at chalk grassland restoration sites. Despite the establishment of similar vegetation to target grasslands, most beetles possessed wide habitat affinities or were associated with common grassland plants (Mortimer *et al.*, 2002). The Hemipteran fauna was also dominated by eurytopic species, and inhabitants of agricultural habitats in the early stages of succession (Morris, 1990).

Predatory beetles and spiders were significantly less abundant on aggregate mounds, although species richness was only greater in the grassland compared

with Thanet mounds in 2015. Species richness was probably similar due to edge effects, although in terms of abundance the change of habitat was apparently a powerful barrier. Similar effects have been demonstrated experimentally for an open-habitat ground beetle in a forest environment (Bertoncelj and Dolman, 2013). Reduced abundance of predatory arthropods is consistent with previous research. Densities of carabids, staphylinids and spiders were lower at unsown plots on earth banks, compared with sown grass cover (Thomas *et al.*, 1991). This was probably caused by enhanced prey availability and amelioration of temperature and humidity in unmanaged grassland (Luff, 1965), creating a favourable environment for arthropod predators.

Abundance of zoophagous beetles showed a similar pattern to predatory beetles and spiders, although this recovered by 2016 on RBF mounds. Greater stone content might explain why abundance was higher on RBF than Thanet mounds in 2016. In dry grassland, plots with experimental manipulation of stone cover had significantly lower beetle species richness than where this represented 50% of the original soil surface (Blight *et al.*, 2011). Stones provided a structural niche for invertebrates, altering microclimate and providing shelter from predators (Sadler *et al.*, 2011). A similar effect was observed for wooden discs placed on the ground to monitor epigeal invertebrates in forest restoration (Bowie and Frampton, 2004). Abundance of carabids and spiders under these microhabitats increased significantly 150–292 days after placement (Bowie and Frampton, 2004). Research in aquatic environments has also demonstrated an effect of substrate size, as with the finding in a freshwater stream riffle that 10–25 mm gravel supported greater abundance and richness of benthic invertebrates than large gravel (> 75 mm), or a mixture of these substrates (Wise and Molles Jr, 1979).

5.4.6 Conclusion and management implications

There were contrasting effects of substrate amendment for the invertebrate groups in this study. The mounds were most attractive to aculeate Hymenoptera in the first year, and nesting activity was limited after spring in the second year by growth of vegetation. Mounds should be created where it is possible to excavate rhizomatous plants, particularly *Cirsium arvense* and *Elytrigia repens* (Jones *et al.*, 2010). Alternatively, these could be maintained by hand weeding,

or combined with a geotextile membrane. An additional factor to consider is the attraction of mounds for burrowing animals (Shaw, 2009b). This provide beneficial ground disturbance (Martinsen *et al.*, 1990) but would affect nest burrows in a small area. Scale is a critical question, although there is a lack of information about the amount of bare ground required to support aculeate populations in combination with foraging resources. Due to the problems of revegetation and disturbance, it might be easier to provide nesting habitat in an enclosed unit. Substrate, compaction, slope angle, and orientation have been identified as important in ecological studies for aculeates (Potts and Willmer, 1997; Heneberg, 2012; Srba and Heneberg, 2012). Future experimental work could include manipulation of these variables.

The communities of beetles and spiders on the mounds resembled existing grassland assemblages. The areas of created habitat were probably too small to have much effect on this, in addition to the limited time available for colonisation and monitoring. Most stenotopic species on the mounds were found in low numbers, although overall abundance did increase in the second year. In general, it is unclear if there is a minimum size or resource requirement below which habitat patches act as sinks, where populations are maintained by continued immigration from more productive areas (Pulliam, 1988). There was no relationship between site area and invertebrate colonisation of restored grasslands from 1.0 ha to 22.8 ha, although population size and resilience of individual species are likely to be higher at larger sites (Woodcock *et al.*, 2015). The value of smaller habitat fragments has been demonstrated for butterflies in Germany, with 50% of species in 0.1 ha as three calcareous grassland fragments, compared to 7.6 ha as a single area (Tscharntke *et al.*, 2002). It is probable that multiple small fragments will encompass greater habitat and habitat-environment heterogeneity (Tscharntke *et al.*, 2002).

The vegetation colonising the mounds was limited to perennial and annual species present in the existing grassland, and additional annuals potentially introduced with the RBF. Phytophagous invertebrates will be unable to colonise created habitat without the presence of suitable host plants and so seed addition should be considered. In arable reversions, floristic diversity and sward structural complexity were the best predictors of grassland specialist invertebrates and high overall invertebrate diversity (Woodcock *et al.*, 2015).

Plants must be sufficiently abundant for location by colonising insects and to subsequently support a viable population (Mortimer *et al.*, 1998). This would increase vegetation cover on the mounds, which emphasises the difficulty of accommodating the requirements of ground-nesting aculeates and phytophagous beetles in a limited space. Larger areas around 400 m^2 are likely to be ecologically valuable (Tscharntke *et al.*, 2002) and practically viable, although it is difficult to generalise results from semi-natural habitat. While it was not possible in the locations for the present study, future research should also investigate the effects of substrate and topography separately, including comparison with local soils.

It is questionable whether the grasslands in this study were suitable locations for habitat creation, as greater benefits might accrue nearer to existing brownfields. Understanding the spatial movement of insects is challenging, given the difficulty of studying dispersal of very small species at the landscape scale. For a metapopulation of *Plebejus argus* in north Wales, colonisation of patches in a single step took place up to 600 m, although most movements were much shorter (Thomas and Harrison, 1992). In a study of grassland restoration, the most successful arable reversions were within 500 m of existing habitat, with no clear threshold to potentially exclude sites (Woodcock *et al.*, 2015). Larger areas of semi-natural grassland within 2 km of reversion sites were not positively correlated with restoration success (Woodcock *et al.*, 2015). Other research has found that some groups may experience greater barriers, as colonisation of heathland indicator Hemiptera declined exponentially across distances up to 40 m (Littlewood *et al.*, 2009). This suggests that habitat creation very close to existing brownfields is most likely to facilitate invertebrate colonisation in the short term.

Larger scale habitat creation has been considered to compensate for development affecting existing sites, particularly early successional and ruderal ecosystems (Morris *et al.*, 2006). While continuing to respond to habitat loss, there may be other opportunities for habitat creation. Given the cost of importing materials, this may be most practical in locations where restoration is already required, such as disused landfills. This could create new areas of nature conservation interest, which may resemble existing ecosystems but without the need to exactly recreate these (Bradshaw, 1977). Recent studies of carabids at

capped landfills restored with topsoil found no species with conservation status (Rahman *et al.*, 2015). Colonisation partly depends on the regional species pool and assemblages were similar to surrounding grassland reference sites. It is also possible that low-nutrient substrates and a diverse plant community could enhance colonisation by stenotopic species (Baranová *et al.*, 2015). Specific measures to benefit early successional invertebrates could be included in other engineering projects, which are usually focused on birds, as with the creation of intertidal habitat with 3 Mt subsoil from Crossrail excavations (Davis and Russell, 2015).

This study has shown the possibility of manipulating substrate for the benefit of invertebrate conservation in an ecologically short time. This is consistent with previous research (Topp *et al.*, 2010; Connop *et al.*, 2014 ; Fortel *et al.*, 2016) but it is unclear whether the field-scale manipulations redistributed existing invertebrate populations within the grasslands, or can form part of landscape scale population dynamics. It was also only possible to detect short-term trends in the two years available for monitoring. Understanding these spatial and temporal elements would be critical for effective implementation of habitat creation, in addition to the design of features. The study of community dynamics in created habitat over longer timescales and at varying distances from existing brownfields is necessary to achieve this.

Chapter 6 – Artificial aculeate nests on green roofs in London

6.1 Introduction

Urbanisation has had complex effects on biodiversity. High species richness in domestic gardens (Thompson *et al.*, 2003) and derelict sites (Small *et al.*, 2003) was probably linked to small-scale habitat heterogeneity. Negative trends in city centres primarily resulted from the large area of artificial impervious surfaces (McKinney, 2008). There is increasing recognition that urban green infrastructure can support biodiversity and associated ecosystem services (Connop *et al.*, 2016; Schwarz *et al.*, 2017), and rooftops are valuable for this in areas of high building density. There was an estimated 3.2 million m^2 of roof space available for conversion in four sample areas of London (Greater London Authority, 2008) and green roofs were part of the strategic plan for integrated economic, environmental and social development in the city (Greater London Authority, 2016). In most systems, plants are established in a light-weight growth medium, above drainage, root barrier and water proofing layers (Williams *et al.*, 2014). Benefits include improved building energy efficiency (Jaffal *et al.*, 2012), cooler microclimates (Castleton *et al.*, 2010) and storm water attenuation (Stovin, 2010). Recent designs also incorporate habitat features characteristic of brownfield sites that are a regional conservation priority (Nash *et al.*, 2016).

There is mixed evidence of the benefits of green roofs for biodiversity (Williams *et al.*, 2014). Colonising plant communities on green roofs in France were principally composed of widespread urban generalists but also included protected species (Madre *et al.*, 2014). Bat activity, assessed by numbers of call sequences and feeding events, was significantly higher over biodiverse roofs compared to conventional buildings (Pearce and Walters, 2012). Invertebrate communities on green roofs included beetles (MacIvor and Lundholm, 2011), bees (Brenneisen, 2006), spiders (Braaker *et al.*, 2014) and soil microarthropods (Rumble and Gange, 2013). Some species had conservation status, including 10% of Coleoptera and Araneae collected on *Sedum* and biodiverse roof plantings in London (Kadas, 2006). Bee species richness and abundance were similar at two green roofs in Toronto, compared with grassy sites at ground level (Colla *et al.*, 2009). Green roofs in Chicago supported native bees but with lower abundance and species richness than areas of park

and prairie vegetation at ground level (Tonietto *et al.*, 2011). In the same city, bee abundance on green roofs was reduced by 50% compared with ground level, and larger species (body size > 12 mm) were poorly represented (Ksiazek *et al.*, 2012). In each case, green roofs were potentially valuable habitats but not exact replicas of sites at ground level.

Potential influences on green roof aculeate diversity include on-roof conditions and the surrounding landscape. In a comparison of six green roofs in Chicago, the greatest number of bee species and individuals was recorded on a roof with the highest plant diversity (Tonietto *et al.*, 2011). Substrate depth may be a critical factor in determining roof vegetation. Fewer species were able to flower and set seed in shallow green roof microcosms at ground level, containing a mixture of expanded clay and sand (Olly *et al.*, 2011). Depth may interact with substrate type, as admixtures of recycled inorganic materials improved plant coverage and species richness (Molineux *et al.*, 2015). The height of a roof might reduce rates of colonisation and genetic outcrossing. Bee abundance was three times greater on a third storey building compared with a fifth storey roof, although it was not possible to separate effects of location and different plantings (Colla *et al.*, 2009). Brood cells in trap nests were less abundant with increasing building height, and the number of unfinished nests increased (MacIvor, 2016). Landscape context includes land use at ground level and conditions on other roofs. In a study of 40 extensively managed green roofs in Zurich, community composition of bees was mainly explained by habitat connectivity, with carabids and spiders more influenced by local environmental conditions (Braaker *et al.*, 2014).

For bees and wasps, habitat quality is related to availability of foraging and nesting resources (Potts *et al.*, 2005). It is unclear whether the life cycles of species recorded as adults on roof tops are completed without access to resources at ground level. Adults of two stem-nesting *Ceratina* were much more abundant at ground level than on green roofs in Toronto, where nesting resources were absent (Colla *et al.*, 2009). Observations are lacking for nesting behaviour of burrowing species in green roof substrates (Tonietto *et al.*, 2011). The distance a species travels between ground and roof level may have implications for reproductive fitness. *Megachile rotundata* females produced fewer offspring with increased flight distance to forage at ground level, and a

greater proportion of sons (Peterson *et al.*, 2006). Some habitat patches could act as sinks for immigration from more productive areas (Pulliam, 1988). From this perspective, a biodiverse roof might represent a partial habitat (Westrich, 1996), with suitable conditions for foraging but not nesting. A mosaic approach to habitat creation suggests that provision of rooftop nesting opportunities could facilitate or maximise populations of aculeate Hymenoptera.

Female cavity nesting aculeates construct brood cells in existing cavities, including hollow plant stems (Bogusch *et al.*, 2015), abandoned insect burrows (Potts *et al.*, 2005) and gaps beneath rocks (Else and Edwards, 1996). Species partition cells with specific materials from the local environment, for example leaves (Strickler *et al.*, 1996), mud (Raw, 1972) and grit (Jenkins and Matthews, 2004). Cavities created by human activities are also exploited, such as nail holes, and cracks in mortar (Raw, 1972). This behaviour has allowed the placement of trap nests, to examine aspects of species and community ecology (MacIvor, 2017). These have been successfully used to investigate time spent foraging (Raw, 1972), distances travelled from nests (Gathmann and Tscharntke, 2002) and resources required for larval development (Praz *et al.*, 2008). Colonisation and diversity in trap nests along environmental gradients have also been studied, to investigate the impacts of landscape change on aculeate communities. These include heavy metal pollution (Moroń *et al.*, 2012), forest canopy cover (Sobek *et al.*, 2009), urbanisation (Fortel *et al.*, 2016) and habitat fragmentation (Coudrain *et al.*, 2014).

Cardboard tubes, drilled wooden blocks, and plant stem internodes are widely available materials for cavity trap nests (MacIvor, 2017). It has also been possible to facilitate species nesting in alternative cavities, such as *Osmia inermis* beneath overturned flower pot saucers, as a surrogate for the underside of stones (Sheffield *et al.*, 2015). In comparison, there is a lack of knowledge about experimental units for burrowing aculeates. Buckets of soil have been used in the laboratory to initiate nest burrows in wood-inhabiting Halictine bees (Barrows, 1973). These have not been used in field experiments, and ecological knowledge of nesting habits has developed from observations of individual bee (Larsson and Franzén, 2007) and wasp species (Field, 1989). Preferences for particle size, compaction and slope angle vary between taxa (Srba and Heneberg, 2012), as well as the depth and arrangement of tunnels and brood

cells. This approach has limitations for community level research, and experimental manipulations of substrate have also relied on in-situ monitoring (Fortel *et al.*, 2016).

A previous study in Canada examined the ecology of cavity nesting aculeates on rooftops in Toronto (MacIvor, 2016). This identified 27 species from trap nests, with increasing building height correlated with reduced abundance of brood cells and greater numbers of unfinished nests. Smaller areas of green space within a 600 m radius of rooftops were also correlated with reduced species richness and abundance. The study did not quantify on-roof floral resources or surrounding aerial green space. In addition, the results cannot necessarily be transferred to other regions. Nineteen native taxa were reared but eight introduced species comprised 73.8% of aculeate abundance (1,601 individuals). There was no attempt to address enhancements for soil-nesting species. This could also be relevant at ground level, where attention has focused on improving foraging resources available to bees, particularly in agri-environment schemes (Potts *et al.*, 2005). Mosaic habitat creation through nesting resource provision could support diverse aculeate communities in urban habitats, providing pollination (Kremen *et al.*, 2004) and predation services (Rogers and Potter, 2003).

This study evaluates colonisation of artificial cavity and substrate nests by aculeate Hymenoptera on roof tops in London. This was monitored in a single year, on biodiverse green roof designs and unvegetated roofs. It was hypothesised that nesting aculeate assemblages would be influenced by a combination of on-roof conditions and landscape context. This was investigated by the following questions: (1) Does building height influence colonisation of roof-top artificial nests by cavity and ground-nesting bees and wasps? (2) Is colonisation of biodiverse roof designs more frequent than unvegetated roofs? (3) Do on-roof conditions, or the area of surrounding greenspace, have a greater influence on assemblages of cavity and ground-nesting bees and wasps on roofs?

6.2 Methods

6.2.1 Site selection

Eighteen roofs within London were available in this study (Figure 6.1). There were 12 biodiverse green roofs, with either wildflower planting, or a *Sedum* matt retrofitted after installation with additional substrates and seeding. There were also six conventional roofs, constructed from asphalt, or concrete roofing tiles. Data on roof heights were not available for all sites, so building levels were used as a proxy for this (Table 6.1). It was not possible to design the experiment to stratify sites among all environmental variables. This meant that the tallest buildings were in central London, where the area of ground level green space was also smallest. The minimum distance between roofs was 100 m, for the biodiverse and conventional roofs at the Victoria and Albert Museum. The cavity nest experimental unit consisted of seven wooden blocks, with drilled holes 3 mm – 9 mm diameter. Each block was 9 cm x 10 cm x 9 cm deep, weighed 420 g, and contained 15 holes of a single size, 7.5 cm deep. These were held in a wooden frame 100 cm above the ground (Figure 6.2). This design ensured that the blocks were at a standard height on the roof, and the order of blocks in each experimental unit was randomised.

Units were installed in early April 2016, facing south. The disadvantage of wooden blocks was that individual nest loss, in which a cell was initiated but no larva developed, could not be assessed. Cardboard tubes and grass internodes have been successfully used for this purpose (MacIvor, 2017). At the time of this study, sufficient sizes of cardboard tubes were unavailable in small quantities in the UK, and grass internodes (Gathmann *et al.*, 1994) were rejected due to the difficulty of standardising hole diameters. The substrate nest experimental unit was a 12 litre round plastic container, 30 cm in diameter at the top, and with drainage holes in the base. This was filled with quarry-dug Thanet sand (Figure 6.2), with particle size distribution 63 μm – 1.18 mm. Units were installed at the same time as the cavity nests, and placed on the bottom of the wooden frame to act as ballast. Units were collected from the roofs in early October 2016, and aculeates were reared in 2017 to determine species richness and abundance for solitary bees and wasps.

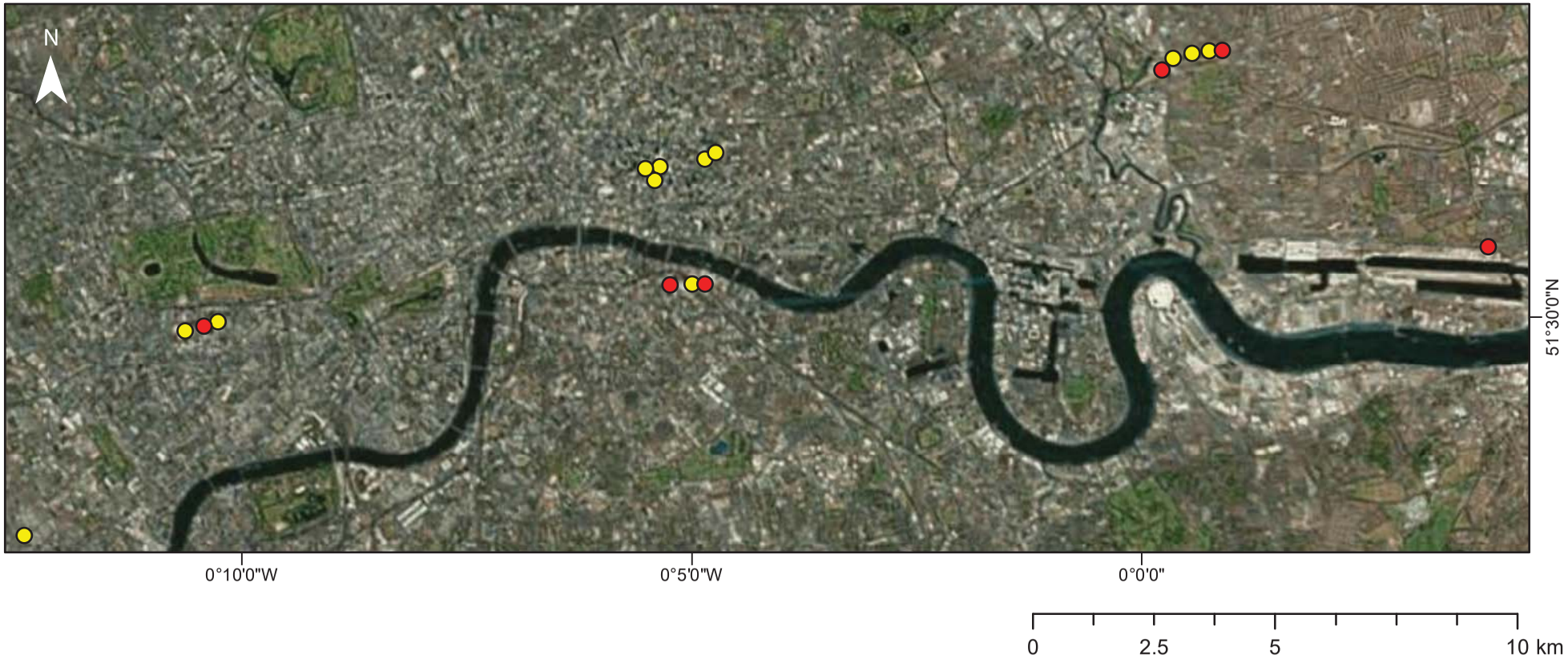

Figure 6.1 Location map for artificial cavity and substrate nests placed on roofs in London, 2016. Biodiverse roofs were either diverse wildflower plantings, or a sedum matt retrofitted after installation, with additional substrates and seeding (n=12). Unplanted roofs were constructed from asphalt, or concrete roofing tiles (n=6). Basemap source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS user community.

Table 6.1 London roofs included in artificial aculeate nest project, 2016. Biodiverse roofs were either wildflower plantings, or a sedum matt retrofitted after installation, with additional substrates and seeding (n=12). Unplanted roofs were constructed from asphalt, or concrete roofing tiles (n=6). Roof height was described categorically as the number of building levels. Vascular plant species on each roof were recorded in late May and early June, with floral resources counted in five 1 m^2 quadrats as the number of individual flowers, or the number of capitula for multiflowered stems.

Site	Grid reference	Planting	Building levels	Plant species richness	Floral resources
201 Bishopsgate	TQ33278188	biodiverse	12	48	1,054
Basinghall Avenue	TQ32588148	biodiverse	10	72	313
Exchange House	TQ33368196	biodiverse	12	30	62
Favart Road, Fulham	TQ25367677	biodiverse	3	63	258
More London	TQ33248012	biodiverse	8	68	516
Museum of London	TQ32158158	biodiverse	5	59	2,245
Natural History Museum	TQ26607902	unplanted	8	3	0
Stratford Westfield	TQ38388460	biodiverse	4	90	480
Stratford Westfield	TQ38368449	unplanted	4	29	9
Tooley Street	TQ33278009	unplanted	3	5	0
UEL Stratford University Square	TQ38998464	biodiverse	4	36	716
UEL Stratford Library	TQ39398478	biodiverse	4	32	155
UEL Stratford	TQ39468477	unplanted	3	0	0
UEL Docklands	TQ43198081	unplanted	3	0	0
Union Street	TQ32468002	unplanted	3	28	17
V&A Museum	TQ27037923	biodiverse	3	56	84
V&A Museum	TQ26957920	unplanted	4	0	0
Wood Street	TQ32348126	biodiverse	8	82	174

Figure 6.2 Artificial aculeate nesting unit for rooftop experiment. The cavity nest comprised seven wooden blocks with 15 drilled holes, 3 mm – 9 mm diameter. The substrate nest was a 12 litre round plastic container, 30 cm in diameter at the top. This was filled with quarry-dug Thanet sand (particle size distribution 63 µm – 1.18 mm).

Figure 6.3 Artificial aculeate nesting unit, designed by John Little (Grass Roof Company, Essex). 1.6 m diameter, constructed from two sheets of 10 mm perforated stainless steel, filled with Thanet sand (particle size distribution 63 µm – 1.18 mm) over a geotextile membrane.

6.2.2 Vascular plants

All vascular plant species on each roof were recorded in late May and early June 2016, during a systematic walk-over, with identifications from Stace (2010). To provide a separate measure of forage availability, the abundance of flowers on all dicotyledon species was counted within five randomly placed 1 m^2 quadrats. Single flowers were counted as one flower 'unit', or in the case of multiflowered stems, the number of capitula (e.g. *Senecio squalidus*) or heads (e.g. *Trifolium dubium*) (Carvell *et al.*, 2007).

6.2.3 Landscape variables

Land use data were not fully compiled due to the lack of reared specimens for subsequent analysis. The intention was to quantify the proportion of managed greenspace and derelict land within buffers at three distances from each roof, 100 m, 500 m and 1 km. These are similar to previous investigations of landscape effects on aculeate populations (Schüepp *et al.*, 2011; MacIvor, 2016), including maximum distances for foraging bees available from literature (Zurbuchen *et al.*, 2010).

6.3 Results

Vegetation varied widely between roofs (Table 6.1). The median plant species richness for biodiverse designs was 59 (minimum 30, maximum 90), and the median floral resources was 155 units (minimum 17, maximum 2,245). The median plant species richness for unplanted roofs was 3 (minimum 0, maximum 29), and the median floral resources was 0 (minimum 0, maximum 9). Forty solitary bees were reared from artificial cavity nests at five roofs, *Osmia bicornis* at four roofs and *Osmia caerulescens* at a fifth location (Table 6.2). At one site where *Osmia bicornis* was reared, and another site without rearing records, there were mud-sealed cavities but no adults emerged. No species were reared from artificial substrate nests.

6.4 Discussion

There was limited colonisation of artificial cavities by aculeate Hymenoptera on roofs in London, with two species and 41 individuals reared. No aculeates were reared from artificial substrate nests. It is likely that colonisation would increase at experimental units installed for multiple years. Significantly larger numbers of

Table 6.2 Sites colonised, hole size utilisation and abundance of each aculeate Hymenoptera species reared in 2017, from artificial wooden block nests set on London rooftops in 2016. For roofs listed without family and taxon, no adults emerged from mud-sealed cavities. Biodiverse roofs were either diverse wildflower plantings, or a sedum matt retrofitted after installation, with additional substrates and seeding (n=12). Unplanted roofs were constructed from asphalt, or concrete roofing tiles (n=6). For locations, see table 6.1.

Family	Taxon	Roof	Cavity (mm)	Total
Megachilidae	*Osmia bicornis*	V&A biodiverse	6	3m
Megachilidae	*Osmia bicornis*	V&A biodiverse	8	15m, 5f
Megachilidae	*Osmia bicornis*	V&A biodiverse	9	2f
Megachilidae	*Osmia bicornis*	V&A unplanted	8	12m, 2f
Megachilidae	*Osmia caerulescens*	More London biodiverse	5	2m
-	-	V&A biodiverse	6	4 cavities
-	-	V&A biodiverse	7	10 cavities
-	-	V&A biodiverse	8	6 cavities
-	-	NHM unplanted	6	2 cavities
-	-	NHM unplanted	8	1 cavity
-	-	More London biodiverse	5	3 cavities

individuals were reared in the second year from artificial cavity nests at ground level (Fortel *et al.*, 2016). Natal site fidelity has been demonstrated for *Osmia bicornis* (=*rufa*), with 80% of all re-observed females in a mark-recapture experiment provisioning around the maternal nest site (Steffan-Dewenter and Schiele, 2004). Short duration experiments with limited capacity for data collection are also vulnerable to disruption by poor weather. June 2016 was the second-wettest month in Essex since records were first collected in 1910 (Met Office Press Office, 2016) and this probably affected aculeate activity. Twenty-six mud-sealed cavities did not produce adults, with no evidence that drilled wooden blocks are unsuitable for larval development (MacIvor, 2017). Damp conditions might have increased larval mortality from fungal growth (Chilcutt and Cowan, 1993).

Populations of roof-top stem-nesting species were a possible limit to colonisation. Plants with pithy or hollow stems were rare, with *Phragmites australis* and *Rubus fruticosus* agg. only recorded on one roof. Abundance of stem-nesting bees in a Mediterranean *Pinus* forest was greatest in areas dominated by dwarf shrubs with pithy stems (Potts *et al.*, 2005). Other environmental variables in the present study were not fully compiled, although some informal observations of the landscape were made during monitoring visits. There was a geographical cluster among colonised rooftops, including three sites in South Kensington – biodiverse (3rd storey) and unvegetated (4th storey) roofs on the Victoria and Albert Museum, and an unvegetated roof at the Natural History Museum (8th storey). This possibly reflected the age of the local building stock, and greater use of brick and mortar as building materials. Fixing holes in brick are frequently used by cavity nesting aculeates, and eroded mortar was a nest site for *Osmia bicornis* (=*rufa*) (Raw, 1972). Recent constructions in central and east London mainly use steel and composites (Taranath, 2016) and probably offer less nesting opportunities.

There were larger areas of established green space around the roofs in South Kensington, including the Oratory Gardens next to the Victoria and Albert Museum green roof, and a wildlife garden below the Natural History Museum unvegetated roof. A previous study in Toronto highlighted the importance of landscape context, as smaller areas of green space around rooftops were

correlated with reduced species richness and abundance of cavity-nesting aculeates (MacIvor, 2016). Ground level green space was also a predictor of abundance and diversity in studies sampling adult bees on vegetated roofs (Tonietto *et al.*, 2011; Braaker *et al.*, 2014). The greatest abundance on the lowest roof is consistent with previous research, as increasing horizontal distance from floral resources resulted in fewer offspring for cavity-nesting bees (Peterson and Roitberg, 2006; Schüepp *et al.*, 2011). There was no apparent effect for on-roof floral resources in the present study. Other research sampling adult bees has identified the greatest diversity at biodiverse green roofs (Brenneisen, 2006; Madre *et al.*, 2013). This suggests that some biodiverse roofs may provide suitable foraging conditions, while lacking structures to support nesting (MacIvor, 2016).

Although there was only anecdotal evidence in the present study, the importance of surrounding habitat quality is a recurrent theme in pollinator research (Steffan-Dewenter *et al.*, 2002). This suggests that sufficient areas of appropriately managed green infrastructure at ground level are critical to maintain urban aculeate populations. It would be of interest to test whether large-scale roll out of rooftop green infrastructure in future developments might enhance nesting density compared with ground level. In both situations, nest sites may be a greater limitation for some species than floral resources (Everaars *et al.*, 2011). Artificial cavity nests could be included routinely on buildings as part of a mosaic approach to habitat creation, although the potential for accumulation of natural enemies is recognised (MacIvor and Packer, 2015), as well as potential impacts of building height on nest completion (MacIvor, 2016) and sex ratio (Peterson *et al.*, 2006). Nesting structures are also a useful tool to increase public awareness of invertebrate conservation (Fortel *et al.*, 2016). There has been limited progress in research with trap nests other than for cavity-nesting aculeates. Colonisation of brick, wooden and tile domiciles by bumblebees was generally low, on average 1.5% from 654 nests set over three years (Fussell and Corbet, 1992).

The present study has added little to understanding of artificial substrate nests, although breeding activity was observed in trials of the experimental unit at ground level, from the solitary wasps *Oxybelus uniglumis*, *Gorytes laticinctus*, and a *Crossocerus* species. Regardless of viability, this only provided a

horizontal surface for colonisation and further development of the design would be valuable. Discussion with a green infrastructure fabricator led to the construction of a cylinder, from lengths of 10 mm perforated stainless steel sheet bolted together (Figure 6.3). This provided horizontal and vertical bare surfaces. The prototype was 1.6 m diameter and 0.6 m high, and was colonised from the horizontal surface by the solitary wasp *Oxybelus uniglumis*, and from the vertical surface by the solitary bee *Lasioglossum morio*, and the solitary wasps *Cerceris rybyensis*, *Anoplius nigerrimus* and *Crossocerus elongatulus*. It is unclear whether there is a minimum size for attraction or successful completion of larval development, although it would be sensible to begin with a portable unit (about 15 kg). Larger features could be installed permanently at ground level, and would be less prone to growth of vegetation or disturbance than unconstrained substrate. Local and imported substrates could be compared, in addition to other experimental manipulations, and in relation to landscape variables.

Chapter 7 – General Discussion

7.1 Overview

Loss and fragmentation of semi-natural habitat, and a reduction in the capacity of the matrix of agricultural land to support biodiversity, are pressing concerns for UK nature conservation. Urban derelict and post-industrial sites now have an important function as surrogates for semi-natural early successional habitats. The UK planning process encourages brownfield redevelopment and while it does recognise the potential for biodiversity value, protection afforded to sites has been limited (Robins *et al.*, 2013). The aim of this research was to assist with conservation of nationally important brownfield invertebrate communities under these circumstances. This was completed within the framework outlined in the 'Lawton Report', which emphasised ecological networks at different scales (Lawton *et al.*, 2010). A key objective was to create practical outputs for land managers, enhancing brownfield invertebrate conservation at existing and new sites. The first three chapters examined restoration and habitat management at Canvey Wick, a core site for brownfield nature conservation. Core areas are important as locations where species can thrive, and disperse to other parts of the network, including restoration areas. The final two chapters concerned small scale habitat creation in the wider landscape, emphasising the potential value of a mosaic approach in design and management for invertebrate conservation.

7.2 Canvey Wick – restoration

7.2.1 Taxonomic analysis

Two areas of closed canopy scrub at Canvey Wick were cleared and scraped to mineral substrate in successive winters after 2014. Ordination revealed distinct groups of carabids, staphylinids and spiders in each restoration area, including stenotopic species. Broad taxonomic coverage was attempted to increase the range of potential responses perceived to management work. It was anticipated that a specialist carabid fauna might not develop in the year after ground disturbance, based on research in dry, sandy grasslands (Pedley *et al.*, 2013; Olsson *et al.*, 2014). In the present study, there were benefits for carabids and staphylinids within a year of scraping. This difference probably resulted from the seasonally wet conditions at Canvey Wick, and underlines the difficulty in

generalising results between different ecosystems. The dredging deposits were reminiscent of dune slacks, although the extensive literature about dune slack restoration is limited to vegetation (Curreli *et al.*, 2013) and soil microarthropods (Jungerius *et al.*, 1995). A distinct invertebrate fauna is associated with fluctuating water levels (Webb and Mott, 2015) and further research is required to establish whether this could benefit from manipulation of topographical and hydrological gradients. This would be valuable at Canvey Wick, and also in natural dune slacks.

The present research was limited to single areas for treatments, so findings should be interpreted with caution, as there was potential to confound management effects and location, even with ecologically plausible results. A further limitation was the short time series available for assessment of the invertebrate communities. Monitoring continued in 2016, although extensive groundwater flooding prevented formal comparison with data for 2015. Only four *Neobisnius procerulus* were recorded, compared with 28 at newly scraped plots in the previous year. It is likely that this species has a critical requirement for exposed substrate (Lott and Anderson, 2011), although this cannot be separated from the effects of weather on the trapping program. Newly recorded Near Threatened carabids were *Amara strenua* and *Acupalpus maculatus*, with one and two individuals respectively. Newly recorded Nationally Scarce spiders were *Xerolycosa nemoralis* and *Sibianor aurocinctus*, each with five individuals. It is possible that some of these species were present at low density in 2015. This also emphasises the dynamic nature of early successional invertebrate communities and the need for greater replication, to increase confidence in inferences from the results.

The source for many early successional species colonising the restoration areas at Canvey Wick was unclear. There may have been some long-distance dispersal, as reference monitoring plots were characterised by a partial assemblage of species associated with drier habitats. The extent to which the restoration work recreated earlier invertebrate interest is also unknown. Previous surveys (Catherine Bickmore Associates, 2004) did not produce equivalent data for comparison with post-restoration communities. From species lists, the halophytic carabids *Anisodactylus poeciloides* and *Dyschirius chalceus* (not formally introduced to the British list at the time of writing, see Luff (2007))

were not recorded in the present survey. There is also anecdotal evidence that halophytic plants have declined, particularly *Aster tripolium* (Harvey, 2012). Only *Salicornia* species were established four years after marine dredgings were deposited at Cliffe Pools, Kent, in contrast to extensive revegetation of the restoration areas at Canvey Wick within one year. This is consistent with experimental disturbance of pulverised fuel ash, which had limited success in recreating the extended early successional phase for vegetation associated with fresh waste (Shaw, 2009a).

7.2.2 Trait-based analysis

The analysis of restoration management at Canvey Wick was successful in describing invertebrate community structure. A limitation was that taxonomic composition is location specific, and did not address the functional traits which mediate responses to disturbance. To address this, species traits for carabids, staphylinids and spiders were analysed from the restoration monitoring samples. There was evidence that the management work resulted in trophic shifts, reduced body size and increased dispersal capability. These conclusions offer to add mechanistic understanding to knowledge of species identity in ecological communities. Carabid and staphylinid responses were consistent with previous research (Ribera *et al.*, 2001; Fournier *et al.*, 2015; García-Tejero and Taboada, 2016), although there was contrasting evidence for spiders (Pedley and Dolman, 2014). Increased abundance of active hunting spiders was predictable after disturbance, but had previously been linked to larger body size, and with no evidence of aerial dispersal (Pedley and Dolman, 2014). These differences probably reflect the location of experiments in forested and open landscapes.

Functional metrics were lower in disturbed areas, although these were favoured by species with conservation status. There were similar findings at grey dunes, compared with later stages of vegetation succession (Schirmel *et al.*, 2012). This suggests that in some situations functional metrics might not be appropriate to judge conservation management outcomes, although these have been demonstrated as better predictors of ecosystem services than taxonomic diversity (Gagic *et al.*, 2015). Other studies have identified a lack of cross-taxon congruence in functional metrics (Spake *et al.*, 2016), emphasising the need for caution about generalisation of taxonomic surrogates. A difference compared

with analysis based on taxonomy was the potential to miss shifts in assemblage composition. Among spiders with conservation status, *Pardosa agrestis* was favoured by scraping, *Trachyzelotes pedestris* was most frequent in reference plots, and *Agraecina striata* was present beneath the tree canopy. There are also broader cultural and moral arguments for conservation of species diversity (Senapathi et al., 2015), and this may be important for long-term stability of ecosystems where it provides functional redundancy (Díaz and Cabido, 2001). From this perspective, trait-based studies were complementary to the taxonomic investigation in this research, and both approaches were valuable.

Categorical traits were derived from literature as a convenient method of classification. As a result, there was limited capacity to directly compare the focal invertebrate groups, beyond possible correlations in broad functional responses. Morphological characteristics measured directly from specimens could be compared between groups where knowledge of species ecology differs widely, such as carabids and staphylinids. Other criticisms are that fourth corner analysis cannot incorporate spatial or phylogenetic correlations, violating the assumption of statistical independence between samples. Organisms with a close evolutionary relationship often share characters, partly explaining similar responses to ecological processes (Kraft *et al.*, 2007). Parallel suites of strategies in distantly related lineages emphasise the possible adaptive significance of these traits (Szefer *et al.*, 2017). Principal coordinate analysis demonstrated phylogenetic covariance for some traits, making it impossible to separate a functional response to disturbance from patterns of phylogenetic clustering. While this is recognised, traits could still be used to predict which species will increase or decrease in abundance for a given environmental change (Townsend and Hildrew, 1994).

7.2.3 Management recommendations from restoration work

General management at Canvey Wick should seek to ensure continuous availability of all stages of succession that support invertebrate conservation interest at the site (Kirby, 1992). Based on substantial differences in assemblages in a single year after restoration, this could include scraping different areas annually, or further scrub clearance. The limitation is that this provides no information about an optimum scale or configuration for individual elements in a habitat mosaic (Staley *et al.*, 2015). Mechanical scrub removal

produced distinct invertebrate communities to reference areas of early successional mosaic habitat. It was also more expensive in 2015 to restore 1 hectare of closed canopy scrub (GBP 20453.50), than to disturb 1 hectare of grassland (GBP 8250.00). This suggests a separate need for early interventional management strategies, based on identification of 'tipping points' towards secondary woodland. To limit shrub colonisation, *Betula pendula* immediately adjacent to disturbed areas should be removed. There is little evidence to suggest effective control of *Galega officinalis* at Canvey Wick, particularly as the plant is abundant on adjacent road verges. Hand-pulling or chemical control are recommended. Salt addition to recreate salinity has also been suggested as a general management prescription, although with concerns about limited persistence and economic feasibility (Shaw, 2011). Other options include manufactured aggregates (Molineux *et al.*, 2015), with potential manipulations including texture, moisture retention and salinity.

7.3 Canvey Wick – ground disturbance

7.3.1 Summary

Scraping and excavator disturbance were investigated as conservation management techniques at a dredging deposit dominated by tall graminoids. The treatments were predicted to produce similar invertebrate communities in the first year. Assemblages at scraped plots were distinct, although disturbed plots were more similar to controls. As only limited comparison of carabids and staphylinids was possible from categorical traits, morphological characters were measured. This offered considerable potential to assess generality in trait responses between these groups. Emergent group analysis suggested functional response groups, potentially complementary to the direct linkage between traits and the environment in fourth corner analysis (Aubin *et al.*, 2009). The benefit of this approach has also been demonstrated in analysis of a spider fauna, with limited knowledge of taxonomy or species ecology (Langlands *et al.*, 2011). The results in the present study are provisional, as there was only time to measure single specimens of each sex. Collective effort to create comprehensive trait databases would reduce this difficulty, as morphological trait data has high re-use value (Homburg *et al.*, 2014; Gossner *et al.*, 2015).

It is interesting to consider the potential difference between the ground disturbance plots (combined treatment area 0.018 ha) and the phase two restoration area (1 ha). These were not directly comparable as the experiments were surveyed in different years, and the sampling intensity was five times greater at the ground disturbance plots. The ground disturbance experiment created a mosaic of disturbed and undisturbed habitat patches, compared with the single large scrape in the phase two restoration. For carabids, there were 34 species shared between the experiments, with 10 species unique to the phase two restoration scrape, and 20 species unique to the ground disturbance treatments. A similar pattern was observed for staphylinids and spiders. This relates to wider debate about the utility for conservation of single large, or multiple small habitat fragments (Simberloff and Abele, 1982).

The spatial subdivision of habitat patches was initially considered in terms of the balance of colonisation and extinction from isolated sites (Simberloff and Abele, 1982). The concept is also relevant for small-scale heterogeneity, potentially accessible for local invertebrate movements, rather than only by dispersal or migration. In a California grassland, plant species richness increased by approximately 40% with experimental subdivision of grazing into 32 units, compared with a single area of 64 m^2 (Quinn and Robinson, 1987). This was partly attributed to the effects of above-ground interspecific competition, with the greater perimeter in subdivided treatments accommodating more late-emerging and low-growing species (Quinn and Robinson, 1987). A review of fragmentation studies found a lack of consistency for species richness and abundance of plants, mammals and invertebrates in relation to fragment size (Debinski and Holt, 2000). It is difficult to understand the scale at which heterogeneity depresses species diversity, with mobility and size class affecting the spatial scale of invertebrate responses to habitat variables (Cole *et al.*, 2010). There is also potential for changing responses to fragmentation over time (Debinski and Holt, 2000).

7.3.2 Management recommendations – ground disturbance

Scraping benefited stenotopic beetles, while generalist spider abundance increased at disturbed plots. At present, both techniques are recommended to restore disturbance dynamics in areas dominated by perennial grasses. Further research is required to establish the benefits of succession at scraped plots

created in consecutive years, compared with excavator disturbance. Reduced abundance of predatory and zoophagous species after scraping emphasised the need to maintain interest associated with established grassland. In the absence of information about a suitable scale for disturbance, caution is advised, to apply management patchily within grassland, and not to larger areas than in the present experiment. There are also practical considerations in the application of different ground disturbance techniques. The time to scrape or turn over 100 m^2 was similar, approximately 20 minutes. Scraping was more expensive, as this required a dumper truck in addition to the excavator. The arisings were removed from the experimental area and formed into a bank. An alternative would be to create smaller banks by each scrape. The effects of this should be investigated as potentially ecologically beneficial, or obstructing future management of the site.

7.4 Habitat creation – aggregate mounds

7.4.1 Summary

In this study, 5-tonne piles of Thanet sand and recycled bulk fill (RBF) were placed in unmanaged grassland as a method of mosaic habitat creation. The results demonstrated potential benefits and limitations of small-scale interventions in the landscape matrix. In year one, the mounds were extensively bare and aculeate Hymenoptera colonised both substrates, with partially overlapping species assemblages. Species richness and community composition of stenotopic beetles and spiders were similar for mounds and grassland, and abundance was reduced on Thanet mounds. In year two, nesting activity of aculeate Hymenoptera mostly ceased after May, as the mound surfaces were entirely shaded by vegetation. Resource life span has been identified as a critical component for metapopulation dynamics in ephemeral habitats, exceeding area or spatial configuration (Keymer *et al.*, 2000). Stenotopic beetles and spiders were more abundant on RBF mounds, although composition remained similar to the grassland. The ruderal flora mostly favoured eurytopic Chrysomelidae, although graminivorous Carabidae also benefitted.

Knowledge of behaviour, movement and resource use by many invertebrates in the landscape matrix is rudimentary. One potential advantage to mobile species

colonising ephemeral habitats is a lack of competition and enemies (Dennis *et al.*, 2013). This might not apply for all taxa, as in the present study, cleptoparasitic *Sphecodes* were prospecting the mounds a week after creation, suggesting that some parasites initially search for likely host nesting locations. The matrix could also impact genetic and physical attributes of individual invertebrates and isolated populations, in turn affecting connectivity between populations in main habitat patches. Reduced wing size and/or thorax musculature were predicted for isolated marginal habitat 'islands' (Dennis *et al.*, 2013). Increasing the number of resource stepping stones might help to counter these evolutionary changes. This was apparent for *Pararge aegeria*, which possessed more elongated wings at desert oases in Morocco, suggesting that these were colonised by a highly dispersive subset of the original population (Habel *et al.*, 2013).

7.4.2 Management recommendations – aggregate mounds

It would be difficult to balance the requirements of aculeate Hymenoptera and stenotopic early successional beetles and spiders at a 5-tonne mound. Most ground-nesting aculeates require unshaded bare substrate (Potts and Willmer, 1997). It might be possible to remove vegetation by hand across half a mound, but this is likely to disturb nest burrows. The potential for disruption of a small area was emphasised at Pitsea, where badgers extensively dug out the Thanet mounds in April 2017. Where it is not possible to create larger areas of bare ground, it is recommended to remove pre-existing and colonising vegetation, and protect exposed substrate from mammals. The performance of local soils should be assessed, as it may be unnecessary to import materials (Fortel *et al.*, 2016). Other terrestrial invertebrates in this study would be better catered for separately, with particular focus on vegetation composition. Establishment of a target floral community would widen the pool of potential colonists (Mortimer *et al.*, 1998). There was limited evidence of colonisation from further afield by stenotopic beetles and spiders. It is unclear whether this was a result of the time available for colonisation, landscape structure, or a threshold minimum size for habitat creation.

7.5 Habitat creation – artificial aculeate nests

7.5.1 Summary

Artificial cavity and substrate nests for aculeate Hymenoptera were placed on biodiverse and unvegetated roofs in London. The aim was to examine the relative importance of on-roof conditions and landscape structure for colonising species. There was limited success in rearing adults from cavity nests, with only 4 of 18 roofs colonised. No adults were bred from substrate nests. This precluded formal analysis of the research questions. Potential problems included poor weather in the survey period, the limited time for exposure of artificial nests, and the pool of species available for colonisation. This should not discourage future development of artificial substrate nests, which have been successful in preliminary trials at ground level. These would benefit from limited risk of disturbance compared to unenclosed substrate, and could be used to test environmental parameters for ground-nesting species including floral resources (Goulson *et al.*, 2015), the effects of pesticides (Woodcock *et al.*, 2017) and landscape structure (Woodcock *et al.*, 2015). This information would be valuable for ecosystem management strategies designed to maintain pollination and predation services (Potts *et al.*, 2003).

The majority of green roof studies have described invertebrate community composition, without addressing the dynamics of colonisation. Experimental creation of nesting habitat could help to develop this type of understanding. The focus on adult occurrence relates to the difficulty of proving breeding for groups such as Coleoptera. The exception was a study of artificial cavity nests for aculeate Hymenoptera in Toronto (MacIvor, 2016), which recorded an average of 25 specimens per year on 29 roofs, for a three-year period with nests removed annually. The present study was less productive, with an average of two specimens in a single year from 18 roofs. Twenty-six cavities on three roofs failed to produce adults, which could have been caused by maladaptation, although this is speculative without longer-term demographic study (Dias, 1996). Where a source-sink structure is produced by passive dispersal, it has been argued that this can depress the density of the source population and potentially endanger its persistence (Dias, 1996). As a result, it is vital that the ecological dynamics of created habitats are understood, in terms of colonisation and connectivity.

Sink habitats are dominated by phenotypes with lower fitness. In Toronto, parasitism was linked to roof planting, and incomplete cells to roof planting and building height (MacIvor, 2016). Sex ratio and resource allocation were not considered, and no data was collected to assess the composition or fitness of populations at ground level. Inclusion of trap nests at ground level and on green roofs would allow comparison of reproductive fitness along a gradient of vertical isolation. Based on previous research, adult emergence dry weight and head capsule width might decrease if limited resources were available, or with increased flight distance to forage, in addition to mothers producing a greater proportion of sons (Peterson *et al.*, 2006). It might also be possible to test the extent to which green roofs function as habitat islands with increasing height (MacArthur and Wilson, 1967), and potential interactions with species-area or individual-area relationships. This type of understanding is vital if appropriate habitats are to be created over suitable spatial scales to support biodiversity conservation targets.

7.6 Assessment of sampling methods

Invertebrates for three studies in this project were collected in pitfall traps, which effectively sampled some of the communities of interest. One consideration was the potential for people to disturb traps, which was a problem in previous inventory surveys at Canvey Wick (Catherine Bickmore Associates, 2004; Harvey, 2012). The site was used for motorbike riding during the present research, so intensive sampling in the restoration areas was completed in three nine-day periods, starting and ending on weekdays. Although no samples were lost, many traps lacked species in the target management groups, and there were insufficient replicates to fit a zero-inflated model (Zuur *et al.*, 2009). Results were pooled within monitoring plots, causing a corresponding loss of replicates from a nested experimental design. It is possible that a smaller number of pitfalls run continuously might have provided sufficient data to analyse at the level of the individual trap, although with greater risk of missing samples if both approaches were not combined.

The biases associated with pitfall trapping are also recognised (Spence and Niemelä, 1994). Although coverage of staphylinids increased the number of species detected, there was considerable trophic and ecological overlap with carabids. Omission of suction sampling failed to fully exploit the potential of

spiders as indicators of vegetation structure (Dennis *et al.*, 2001). Research for grassland invertebrate conservation has successfully used multiple groups to assess management effects (Morris, 2000). In calcareous grassland, grazing produced contrasting effects on leaf-mining Lepidoptera through plant species composition, and spiders by plant architecture (Gibson *et al.*, 1992). In general, groups chosen for study should include contrasting trophic and ecological strategies (Woodcock *et al.*, 2003). In this respect, the pilot study of Diptera utilising the scrub-grassland interface was a valuable counterpart to surveys of the epigeal fauna. Other potentially useful groups for assessment of scrub or woodland include phytophagous Hemiptera (Helden *et al.*, 2012) and Lepidoptera (Goodwin *et al.*, 2017).

Additional sampling methods or replicates would require more resources for data collection and specimen identification. Limiting taxonomic coverage to a smaller number of indicator groups is an obvious way to mitigate for this. Caution is required, as different outcomes were reported for carabids compared with other invertebrates classified by order (Cameron and Leather, 2012b). It is also undesirable to collect a large volume of material that is never identified, which would increase with additional bulk collection techniques. Where taxonomic expertise is unavailable, DNA barcoding (Yu *et al.*, 2012) and metagenomics (Tang *et al.*, 2015) might offer an alternative to process samples. A taxonomically validated reference set of DNA sequences is available for over 90% of British bee species (Carvell *et al.*, 2016) and coverage for other groups is expected to improve in future. This process could classify females and larval collections that cannot be identified from morphology. Count data can be obtained from individual-based DNA barcoding, although costs scale with specimen numbers (Shokralla *et al.*, 2015). Mitogenomic data sets provide estimates of species counts, with costs increasing with the number of samples (Tang *et al.*, 2015).

7.7 Future research priorities

There is considerable need for further research to benefit invertebrate conservation within Open Mosaic Habitat, and to develop understanding of the ecology of species associated with small-scale habitat mosaics. This project has addressed the invertebrate communities associated with bare ground creation, and the comparative effectiveness of management interventions.

Mosaic habitat elements also include floral resources, sward structural heterogeneity, topographic variation, ephemeral waterbodies, and scrub. The effects of patch size and spatial configuration should be assessed for each element, and in relation to other elements where species require these in combination. Comparing individual mosaic elements would have the benefit of optimising sampling techniques for associated invertebrate communities. One of the difficulties in sampling for this research was the difference in effectiveness of methods between habitats. Phytophagous species in the grassland field layer can be collected in suction samples (Brook *et al.*, 2008) but this was impossible in areas of bare sand, or dense leaf litter. Data could be standardised to account for increased sampling effort in larger elements, such as the mean average of multiple samples (Cameron and Leather, 2012a), or adjusted based on species accumulation curves (Helden *et al.*, 2012).

7.7.1 Mosaic element size

Studies to date have usually addressed area at a site level rather than for individual mosaic elements (Lenda *et al.*, 2012; Lönnberg and Jonsell, 2012), although some evidence is available about bare ground patch size. Carabid species richness, abundance and diversity were greatest at 5 m^2 scrapes in heathland, except for 10 m^2 patches with grass tussocks at the perimeter (Cameron and Leather, 2012a). 108 bare ground scrapes were created by turf-stripping heathland at Iping Common, West Sussex, between 2004/05 and 2012/13 (Lyons and Pryor, 2014). These were part of a reintroduction programme for the carabid *Cicindela sylvatica* and covered 0.67 ha, with 90 between 10 m^2 and 50 m^2, and the largest 985.80 m^2. The solitary wasp *Ammophila pubescens* (Nationally Scarce B), its parasite bee-fly *Thyridanthrax fenestratus* (RDB3), the carabid *Anisodactylus nemorivagus* (Nationally Scarce A) and the spiders *Talavera pratensis*, *Aelurillus v-insignitus* and *Xerolycosa nemoralis* (Nationally Scarce B) were recorded using the scrapes. The scrapes with the highest invertebrate species richness were 286.30 m^2 and 466.80 m^2. This suggests that future research should include bare ground creation at scales up to 25 m^2, although sufficient replication may not be possible in a brownfield setting because of variability in substrates, and the small scale of sites.

7.7.2 Mosaic element geometry

Other untested theories concern the shape of mosaic elements. Bare ground creation for heathland invertebrate conservation has been part of a series of research projects, including the National Lowland Heathlands Programme (Edwards, 1994; 1995), Surrey Heathland Project (Edwards, 1996) and West Sussex Heathlands Project (Edwards, 1997; 1998). Early suggestions were to create elongate bare ground patches (Edwards, 1994). Later, the need for wide rather than narrow areas was emphasised, to slow overgrowing vegetation (Edwards, 1998). Other studies have identified a persistent line of bare ground at the edge of scrapes, attributed to slower growth of heather next to established plants (Lyons and Pryor, 2014). To maximise this effect, it was suggested that scrapes should be sinuous and placed a few metres apart. The provisional status of knowledge to benefit invertebrate conservation is evident, and results may be contingent on vegetation communities. Effects of shape may also differ for each mosaic element, and between invertebrate groups. Shape, edge length, and orientation in woodland patches were all found to seasonally affect distribution and abundance of the hoverfly *Episyrphus balteatus* (Sarthou *et al.*, 2005).

7.7.3 Spatial configuration of mosaic habitat elements

The spatial configuration of mosaic habitat elements requires investigation. The effects of isolation have been demonstrated experimentally for a community of endophagous insects associated with *Artemisia vulgaris* (Denys and Schmidt, 1998). Species richness of herbivores, predators, and particularly parasitoids, decreased along a gradient of urbanisation (Denys and Schmidt, 1998). Species richness was reduced at more isolated sites, although overall abundance was not affected until isolation was extreme, due to the abundance of several eurytopic herbivores (Denys and Schmidt, 1998). At the landscape scale, 10 ha of calcareous grassland was predicted to support 98% of butterfly species sampled when composed of about 30 small fragments, compared with 50-60% of species within one or two large grasslands (Steffan-Dewenter and Tscharntke, 2000). The surrounding landscape also has the potential to alter invertebrate responses to smaller scale habitat variables (Weigel *et al.*, 2003). Abundance of the butterfly *Euphydryas aurinia* was positively correlated with sward height (Botham *et al.*, 2011). The optimal sward height varied with

landscape context, from 15 cm in unfragmented landscapes, to 20 cm at more isolated sites (Botham *et al.*, 2011).

The importance of spatial configuration extends to multiple mosaic elements, as many species require these in combination. Tests including diverse habitat structures are lacking (Staley *et al.*, 2015), although information about invertebrate activity ranges provides an indication of the spatial scale at which individual species require different mosaic elements. Within-season foraging tends to occur at smaller scales than dispersal events (Staley *et al.*, 2015). The butterfly *Aricia artaxerxes* moved less than 100 m in 98% of observations, although up to 200 m to find new nectar sources (Ravenscroft and Warren, 1996). While maximum foraging distances for *Hylaeus punctulatissimus* (a small solitary bee) and *Hoplitis adunca* (a larger solitary bee) were 1,100 m and 1,400 m, 50% of females did not forage beyond 225 m and 300 m respectively (Zurbuchen *et al.*, 2010). The spatial distribution of resources can impact species ecology, as *Megachile rotundata* females produced fewer offspring with increased flight distances to forage, and a greater proportion of sons (Peterson et al., 2006).

7.7.4 Studies of mosaic habitat elements

The study of Open Mosaic Habitat should continue with correlative surveys of mosaic habitat elements, and manipulative field experiments. Correlative surveys would be the easiest way to investigate individual mosaic elements, for the variables in sections 7.7.1 – 7.7.3. These have potential to provide rich information about invertebrate assemblages, as increasing sample size would include more sites, and potential habitat and habitat-environment heterogeneity (Tscharntke *et al.*, 2002). This is particularly relevant for elements which take longer to establish, such as scrub. Invertebrates, vegetation composition, small areas of scrub, and topographic heterogeneity should be assessed by field survey. It might be possible to map the position and extent of larger mosaic elements by drone remote sensing, including scrub and scattered trees (Staley *et al.*, 2015). Landscape heterogeneity at a range of scales would be assessed from classifications in the most current UK land cover map (Morton *et al.*, 2011). A limitation with correlative studies is separation of causal mechanisms because of varied site histories, including age and management practices (Woodcock *et al.*, 2015). The results should be used to test experimentally how

management interventions could alter the structure of habitat mosaics, and potentially viable scales.

7.7.5 Experimental Open Mosaic Habitat creation

In addition to management of existing sites, the increasing fragmentation of Open Mosaic Habitat (Robins *et al.*, 2013) must be addressed through experimental habitat creation. Scale should be taken from correlative surveys, although based on a study of calcareous grassland fragments (Tscharntke *et al.*, 2002), a likely minimum area is 400 – 900 m^2. This would allow between four and nine 10 m x 10 m experimental plots. The unmanaged grassland sites in the present research had value for invertebrate conservation, and habitat creation should probably be targeted on brownfields with low conservation value, such as hard standing, on arable land, or landfill restoration. This overlaps with restorations to create species-rich grassland, which is an important part of Open Mosaic Habitat. Arable restorations are sometimes preceded by topsoil removal to reduce phosphorous levels (Woodcock *et al.*, 2015), and could subsequently include substrate addition and topographic modification. The development of vegetation has been manipulated by substrate in a 4-acre garden in Langdon Hills, Essex, including sand, crushed ceramics, mixed concrete and brick, and pure concrete, on a geotextile membrane. There is also a possible benefit for epigeal invertebrates to varied size distribution of some aggregates (Sadler *et al.*, 2011).

Three substrates could be tested against local soil within 400 m^2. The experimental plots could be larger within 900 m^2, or include manipulation of topography in a factorial design. Vegetation and invertebrate communities were linked in grassland restoration, and seed introduction is strongly recommended (Woodcock *et al.*, 2015). Target vegetation communities may not correspond with semi-natural habitats (Bodsworth *et al.*, 2005), particularly ruderal species which are potentially important as a source of pollen and nectar (Connop, 2007). Some structural elements are also required, including dead wood with holes for cavity nesting aculeates. The introduction of invertebrate communities is rarely considered (Mortimer *et al.*, 2002), although for some species threshold distances for colonisation are likely to be a limitation (Gutiérrez *et al.*, 2001). Research is required to develop approaches for artificial transfers that are ecologically effective and economically viable (Connop, 2012). Replicates of

created habitat should be established at different distances from existing Open Mosaic or semi-natural habitats. This would require manipulations over large areas and long-time scales (5 – 10 years) but should improve understanding of the interactions between site management and the surrounding landscape.

7.8 Partnership approach to research

This research was established as a partnership between University of East London and Buglife, a third sector charity dedicated to the conservation of UK invertebrates. It was funded as a collaborative award, based on the Natural Environment Research Council CASE studentship model. The initial requirement for the work originated from discussions between UEL and the non-academic partner, identifying the need for information about novel brownfield conservation management techniques. The process of co-creation embedded impact throughout the research, integrating the practical requirements of reserve management with theoretical and experimental aspects of ecology (Morris, 2000). The results of the project have been presented at Canvey Wick management steering group meetings, and during informal site visits. These will be incorporated into future management prescriptions for the site. The research also received funding from direct partnership with Natural England, and in combination with other stakeholders through the Greater Thames Marshes Nature Improvement Area. This provided the opportunity for habitat creation in areas not covered by conventional conservation management agreements. These contributions to wider ecological networks are also critical for the conservation of biodiversity at a landscape scale.

Chapter 8 – References

'Council Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives' (2008) *Official Journal* L312, pp. 3–30.

Ackland, D.M. (2012) *Draft keys to British Anthomyiidae*. Unpublished manuscript.

Adams, D., De Sousa, C. and Tiesdell, S. (2010) 'Brownfield development: a comparison of North American and British approaches', *Urban Studies*, 47(1), pp. 75-104. doi: 10.1177/0042098009346868

Allan, R.L., Dickinson, G., Dickinson, J.H., Duncan, H.J., Murphy, K.J., Pulford, I.D., Rogerson, R. and Watson, K. (1997) *The natural heritage interest of bings (waste tips) in Scotland: Inventory and review*. Available at: www.snh.org.uk/pdfs/publications/review/048.pdf (Accessed: 9 October 2017).

Andelman, S.J. and Fagan, W.F. (2000) 'Umbrellas and flagships: efficient conservation surrogates or expensive mistakes?', *Proceedings of the National Academy of Sciencies of the USA*, 97(11), pp. 5954-5959. doi: 10.1073/pnas.100126797

Andersen, J. (1983) 'The habitat distribution of species of the tribe Bembidiini (Coleoptera, Carabidae) on banks and shores in northern Norway', *Notulae Entomologicae,* 63(3), pp. 131-142.

(1985) 'Ecomorphological adaptations of riparian Bembidiini species (Coleoptera: Carabidae)', *Entomologia Generalis*, 11(1-2), pp. 41-46. doi: 10.1127/entom.gen/11/1985/41

(1995) 'A comparison of pitfall trapping and quadrat sampling of Carabidae (Coleoptera) on river banks', *Entomologica Fennica,* 6(2-3), pp. 65-77.

Andersen, J. (2000) 'What is the origin of the carabid beetle fauna of dry, anthropogenic habitats in western Europe?', *Journal of Biogeography*, 27(4), pp. 795-806. doi: 10.1046/j.1365-2699.2000.00452.x

Andersen, J. (2011) 'Winter quarters of wetland ground beetles (Coleoptera, Carabidae) in South Scandinavia', *Journal of Insect Conservation*, 15(6), pp. 799-810. doi: 10.1007/s10841-011-9378-7

Andersen, J. and Arneberg, P. (2016) 'Hand collection as a method for assessing the community structure of carabid beetles', *Pedobiologia*, 59(3), pp. 73-81. doi: 10.1016/j.pedobi.2016.02.003

Anderson, A., Carnus, T., Helden, A.J., Sheridan, H. and Purvis, G. (2013) 'The influence of conservation field margins in intensively managed grazing land on

communities of five arthropod trophic groups', *Insect Conservation and Diversity*, 6(3), pp. 201-211. doi: 10.1111/j.1752-4598.2012.00203.x

Anderson, E.C. and Minor, E.S. (2017) 'Vacant lots: an underexplored resource for ecological and social benefits in cities', *Urban Forestry & Urban Greening*, 21, pp. 146-152. doi: 10.1016/j.ufug.2016.11.015

Angold, P.G., Sadler, J.P., Hill, M.O., Pullin, A., Rushton, S., Austin, K., Small, E., Wood, B., Wadsworth, R., Sanderson, R. and Thompson, K. (2006) 'Biodiversity in urban habitat patches', *Science of the Total Environment*, 360(1-3), pp. 196-204. doi: 10.1016/j.scitotenv.2005.08.0358

Arens, S.M. and Geelen, L.H.W.T. (2006) 'Dune landscape rejuvenation by intended destabilisation in the Amsterdam water supply dunes', *Journal of Coastal Research*, 22(5), pp. 1094-1107. doi: 10.2112/04-0238.1

Ash, H.J., Gemmell, R.P. and Bradshaw, A.D. (1994) 'The introduction of native plant species on industrial waste heaps: a test of immigration and other factors affecting primary succession', *Journal of Applied Ecology*, 31(1), pp. 74-84. doi: 10.2307/2404600

Assing, V. and Schülke, M. (2012) *Freude–Harde–Lohse–Klausnitzer: Die Käfer Mitteleuropas: Band 4: Staphylinidae I.* 2nd edn. Heidelberg: Spektrum Akademischer Verlag.

Aubin, I., Ouellette, M.H., Legendre, P., Messier, C. and Bouchard, A. (2009) 'Comparison of two plant functional approaches to evaluate natural restoration along an old-field – deciduous forest chronosequence', *Journal of Vegetation Science*, 20(2), pp. 185-198. doi: 10.1111/j.1654-1103.2009.05513.x

Aubin, I., Venier, L., Pearce, J. and Moretti, M. (2013) 'Can a trait-based multi-taxa approach improve our assessment of forest management impact on biodiversity?', *Biodiversity and Conservation*, 22(12), pp. 2957-2975. doi: 10.1007/s10531-013-0565-6

Audisio, P. (1993) *Coleoptera Nitidulidae-Kateretidae.* Bolonga: Edizioni Calderini. Fauna d'Italia 32.

Awadhwal, N.K. and Thierstein, G.E. (1985) 'Soil crust and its impact on crop establishment: a review', *Soil and Tillage Research*, 5(3), pp. 289-302. doi: 10.1016/0167-1987(85)90021-2

Baars, M.A. (1979) 'Catches in pitfall traps in relation to mean densities of carabid beetles', *Oecologia*, 41(1), pp. 25-26. doi: 10.1007/BF00344835

Babin-Fenske, J. and Anand, M. (2010) 'Terrestrial insect communities and the restoration of an industrially perturbed landscape: assessing success and surrogacy', *Restoration Ecology*, 18(S1), pp. 73-84. doi: 10.1111/j.1526-100X.2010.00665.x

Bakker, E.S. and Olff, H. (2003) 'Impact of different-sized herbivores on recruitment opportunities for subordinate herbs in grasslands', *Journal of Vegetation Science*, 14(4), pp. 465-474. doi: 10.1111/j.1654-1103.2003.tb02173.x

Bakker, J.P. and Berendse, F. (1999) 'Constraints in the restoration of ecological diversity in grassland and heathland communities', *Trends in Ecology & Evolution*, 14(2), pp. 63-68. doi: 10.1016/S0169-5347(98)01544-4

Balisky, A.C. and Burton, P.J. (1995) 'Root-zone soil temperature variation associated with microsite characteristics in high-elevation forest openings in the interior of British Columbia', *Agricultural and Forest Meteorology*, 77(1), pp. 31-54. doi: 10.1016/0168-1923(95)02235-P

Balmer, O. and Erhardt, A. (2000) 'Consequences of succession on extensively grazed grasslands for central European butterfly communities: rethinking conservation practices', *Conservation Biology*, 14(3), pp. 746-757. doi: 10.1046/j.1523-1739.2000.98612.x

Balog, A., Mehrparvar, M. and Weisser, W.W. (2013) 'Polyphagous predatory rove beetles (Coleoptera: Staphylinidae) induce winged morphs in the pea aphid Acyrthosiphon pisum (Hemiptera: Aphididae)', *European Journal of Entomology*, 110(1), pp. 153-157. doi: 10.14411/eje.2013.021

Banaszak, J. and Twerd, L. (2010) 'High number of cuckoo wasps (Hymenoptera: Chrysididae) in areas directly affected by lime and sodium industry', *Polish Journal of Entomology,* 79(3), pp. 291-305.

Baranová, B., Manko, P. and Jászay, T. (2015) 'Waste dumps as local biodiversity hotspots for soil macrofauna and ground beetles (Coleoptera: Carabidae) in the agricultural landscape', *Ecological Engineering*, 81, pp. 1-13. doi: 10.1016/j.ecoleng.2015.04.023

Barbaro, L. and van Halder, I. (2009) 'Linking bird, carabid beetle and butterfly life-history traits to habitat fragmentation in mosaic landscapes', *Ecography*, 32(2), pp. 321-333. doi: 10.1111/j.1600-0587.2008.05546.x

Barrows, E.M. (1973) 'Soil nesting by wood-inhabiting halictine bees, Augochlora pura and Lasioglossum coeruleum (Hymenoptera: Halictidae)', Journal of the Kansas Entomological Society, 46(4), pp. 496-499. Available at: http://www.jstor.org.libezproxy.open.ac.uk/stable/25082599 (Accessed: 8 October 2017).

Barton, P.S., Gibb, H., Manning, A.D., Lindenmayer, D.B. and Cunningham, S.A. (2011) 'Morphological traits as predictors of diet and microhabitat use in a diverse beetle assemblage', *Biological Journal of the Linnean Society*, 102(2), pp. 301-310. doi: 10.1111/j.1095-8312.2010.01580.x

Bauer, L.J. (1989) 'Moorland beetle communities on limestone `habitat islands'. II. Flight activity, and its influence on local staphylinid diversity', *Journal of Animal Ecology*, 58(3), pp. 1099-1113. doi: 10.2307/5144

Bauer, T., Desender, K., Morwinsky, T. and Betz, O. (1998) 'Eye morphology reflects habitat demands in three closely related ground beetle species (Coleoptera: Carabidae)', *Journal of Zoology*, 245(4), pp. 467-472. doi: 10.1111/j.1469-7998.1998.tb00121.x

Baum, K.A., Haynes, K.J., Dillemuth, F.P. and Cronin, J.T. (2004) 'The matrix enhances the effectiveness of corridors and stepping stones', *Ecology*, 85(10), pp. 2671-2676. doi: 10.1890/04-0500

Bell, J.R., Bohan, D.A., Shaw, E.M. and Weyman, G.S. (2005) 'Ballooning dispersal using silk: world fauna, phylogenies, genetics and models', *Bulletin of Entomological Research*, 95(2), pp. 69-114. doi: 10.1079/BER2004350

Belshaw, R. (1993) *Tachinid flies: Diptera: Tachinidae.* London: Royal Entomological Society of London. Handbooks for the Identification of British Insects Vol. 10, Part 4a(i).

Benefer, C., Andrew, P., Blackshaw, R., Ellis, J. and Knight, M. (2010) 'The spatial distribution of phytophagous insect larvae in grassland soils', *Applied Soil Ecology*, 45(3), pp. 269-274. doi: 10.1016/j.apsoil.2010.05.002

Beneš, J., Kepka, P. and Konvička, M. (2003) 'Limestone quarries as refuges for European xerophilous butterflies', *Conservation Biology*, 17(4), pp. 1058-1069. doi: 10.1046/j.1523-1739.2003.02092.x

Benton, T.G., Vickery, J.A. and Wilson, J.D. (2003) 'Farmland biodiversity: is habitat heterogeneity the key?', *Trends in Ecology & Evolution*, 18(4), pp. 182-188. doi: 10.1016/S0169-5347(03)00011-9

Berggren, Å., Birath, B. and Kindvall, O. (2002) 'Effect of corridors and habitat edges on dispersal behavior, movement rates, and movement angles in Roesel's bush-cricket (*Metrioptera roeseli*)', *Conservation Biology*, 16(6), pp. 1562-1569. doi: 10.1046/j.1523-1739.2002.01203.x

Bertoncelj, I. and Dolman, P.M. (2013) 'The matrix affects trackway corridor suitability for an arenicolous specialist beetle', *Journal of Insect Conservation*, 17(3), pp. 503-510. doi: 10.1007/s10841-012-9533-9

Bettacchioli, G., Taormina, M., Bernini, F. and Migliorini, M. (2012) 'Disturbance regimes in a wetland remnant: implications for trait-displacements and shifts in the assemblage structure of carabid beetles (Coleoptera: Carabidae)', *Journal of Insect Conservation*, 16(2), pp. 249-261. doi: 10.1007/s10841-011-9412-9

Betz, O. (1996) 'Function and evolution of the adhesion-capture apparatus of *Stenus* species (Coleoptera, Staphylinidae)', *Zoomorphology*, 116(1), pp. 15-34. doi: 10.1007/bf02526926

Betz, O. (1998) 'Life forms and hunting behaviour of some Central European *Stenus* species (Coleoptera, Staphylinidae)', *Applied Soil Ecology*, 9(1), pp. 69-74. doi: 10.1016/S0929-1393(98)00056-0

Betz, O. and Mumm, R. (2001) 'The predatory legs of *Philonthus marginatus* (Coleoptera, Staphylinidae): functional morphology and tarsal ultrastructure', *Arthropod Structure & Development*, 30(2), pp. 77-97. doi: 10.1016/S1467-8039(01)00029-9

Biederman, L.A. and Whisenant, S.G. (2011) 'Using mounds to create microtopography alters plant community development early in restoration', *Restoration Ecology*, 19(101), pp. 53-61. doi: 10.1111/j.1526-100X.2010.00670.x

Bivand, R. (2017) spdep: spatial dependence: weighting schemes, statistics and models (Version 0.6-15) [R package]. Available at: https://cran.r-project.org/package=spdep (Accessed: 13 October 2017).

Bivand, R. and Piras, G. (2015) 'Comparing implementations of estimation methods for spatial econometrics', *Journal of Statistical Software*, 63(18), pp. 1-36. doi: 10.18637/jss.v063.i18

Bivand, R.S., Hauke, J. and Kossowski, T. (2013) 'Computing the Jacobian in Gaussian spatial autoregressive models: an illustrated comparison of available methods', *Geographical Analysis*, 45(2), pp. 150-179. doi: 10.1111/gean.12008

Blake, S., Foster, G.N., Eyre, M.D. and Luff, M.L. (1994) 'Effects of habitat type and grassland management practices on the body size distribution of carabid beetles', *Pedobiologia,* 38(6), pp. 502-512.

Blandenier, G. (2009) 'Ballooning of spiders (Araneae) in Switzerland: general results from an eleven-year survey', *Arachnology*, 14(7), pp. 308-316. doi: 10.13156/arac.2009.14.7.308

Blight, O., Fadda, S., Orgeas, J., Ponel, P., Buisson, E. and Dutoit, T. (2011) 'Using stone cover patches and grazing exclusion to restore ground-active beetle communities in a degraded pseudo-steppe', *Journal of Insect Conservation*, 15(4), pp. 561-572. doi: 10.1007/s10841-010-9358-3

Bobbink, R. and Willems, J.H. (1987) 'Increasing dominance of *Brachypodium pinnatum* (L.) Beauv. in chalk grasslands: a threat to a species-rich ecosystem', *Biological Conservation*, 40(4), pp. 301-314. doi: 10.1016/0006-3207(87)90122-4

Bodsworth, E., Shepherd, P. and Plant, C. (2005) *Exotic plant species on brownfield land: their value to invertebrates of nature conservation importance*. Available at: http://publications.naturalengland.org.uk/publication/102007 (Accessed: 9 October 2017).

Boesch, D.F. (1977) *Application of numerical classification in ecological investigations of water pollution.* Corvallis, OR: United States Environmental Protection Agency.

Bogusch, P., Astapenková, A. and Heneberg, P. (2015) 'Larvae and nests of six aculeate Hymenoptera (Hymenoptera: Aculeata) nesting in reed galls induced by *Lipara* spp. (Diptera: Chloropidae) with a review of species recorded', *PLOS ONE*, 10(6), e0130802 (23pp). doi: 10.1371/journal.pone.0130802

Bogusch, P., Macek, J., Janšta, P., Kubík, Š., Řezáč, M., Holý, K., Malenovský, I., Baňař, P., Mikát, M., Astapenková, A. and Heneberg, P. (2016) 'Industrial and post-industrial habitats serve as critical refugia for pioneer species of newly identified arthropod assemblages associated with reed galls', *Biodiversity and Conservation*, 25(5), pp. 827-863. doi: 10.1007/s10531-016-1070-5

Bohac, J. (1999) 'Staphylinid beetles as bioindicators', *Agriculture, Ecosystems & Environment*, 74(1), pp. 357-372. doi: 10.1016/S0167-8809(99)00043-2

Bohan, D.A., Boursault, A., Brooks, D.R. and Petit, S. (2011) 'National-scale regulation of the weed seedbank by carabid predators', *Journal of Applied Ecology*, 48(4), pp. 888-898. doi: 10.1111/j.1365-2664.2011.02008.x

Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Stevens, M.H.H. and White, J.-S.S. (2009) 'Generalized linear mixed models: a practical guide for ecology and evolution', *Trends in Ecology & Evolution,* 24(3), pp. 127-135.

Bonte, D., Baert, L. and Maelfait, J.-P. (2002) 'Spider assemblage structure and stability in a heterogeneous coastal dune system (Belgium)', *Journal of Arachnology*, 30(2), pp. 331-343. doi: 10.1636/0161-8202(2002)030[0331:SASASI]2.0.CO;2

Bonte, D., Lens, L. and Maelfait, J.-P. (2006) 'Sand dynamics in coastal dune landscapes constrain diversity and life-history characteristics of spiders', *Journal of Applied Ecology*, 43(4), pp. 735-747. doi: 10.1111/j.1365-2664.2006.01175.x

Bonte, D., Lens, L., Maelfait, J.-P., Hoffmann, M. and Kuijken, E. (2003a) 'Patch quality and connectivity influence spatial dynamics in a dune wolfspider', *Oecologia*, 135(2), pp. 227-233. doi: 10.1007/s00442-003-1195-6

Bonte, D., Maelfait, J.-P. and Hoffmann, M. (2000) 'The impact of grazing on spider communities in a mesophytic calcareous dune grassland', *Journal of Coastal Conservation*, 6(2), pp. 135-144. doi: 10.1007/BF02913810

Bonte, D., Vandenbroecke, N., Lens, L. and Maelfait, J.-P. (2003b) 'Low propensity for aerial dispersal in specialist spiders from fragmented landscapes', *Proceedings of the Royal Society of London B: Biological Sciences*, 270(1524), pp. 1601-1607. doi: 10.1098/rspb.2003.2432

Borcard, D., Legendre, P. and Drapeau, P. (1992) 'Partialling out the spatial component of ecological variation', *Ecology*, 73(3), pp. 1045-1055. doi: 10.2307/1940179

Bossuyt, B., Butaye, J. and Honnay, O. (2006) 'Seed bank composition of open and overgrown calcareous grassland soils - a case study from Southern Belgium', *Journal of Environmental Management*, 79(4), pp. 364-371. doi: 10.1016/j.jenvman.2005.08.005

Botham, M.S., Ash, D., Aspey, N., Bourn, N.A.D., Bulman, C.R., Roy, D.B., Swain, J., Zannese, A. and Pywell, R.F. (2011) 'The effects of habitat fragmentation on niche requirements of the marsh fritillary, *Euphydryas aurinia*, (Rottemburg, 1775) on calcareous grasslands in southern UK', *Journal of Insect Conservation*, 15(1-2), pp. 269-277. doi: 10.1007/s10841-010-9344-9

Boulet, M.P. and Larocque, A.C.L. (1998) 'A comparative mineralogical and geochemical study of sulfide mine tailings at two sites in New Mexico, USA', *Environmental Geology*, 33(2), pp. 130-142. doi: 10.1007/s002540050233

Bowie, M.H. and Frampton, C.M. (2004) 'A practical technique for non-destructive monitoring of soil surface invertebrates for ecological restoration programmes', *Ecological Management & Restoration*, 5(1), pp. 34-42. doi: 10.1111/j.1442-8903.2004.00171.x

Bowstead, S. (1999) *A revision of the Corylophidae (Coleoptera) of the West Palaearctic region.* Genève, Switzerland: Muséum d'histoire naturelle.

Braaker, S., Ghazoul, J., Obrist, M.K. and Moretti, M. (2014) 'Habitat connectivity shapes urban arthropod communities: the key role of green roofs', *Ecology*, 95(4), pp. 1010-1021. doi: 10.1890/13-0705.1

Bradshaw, A.D. (1977) 'Conservation problems in the future', *Proceedings of the Royal Society of London B: Biological Sciences*, 197(1126), pp. 77-96. doi: 10.1098/rspb.1977.0058

(1983) 'The reconstruction of ecosystems: presidential Address to the British Ecological Society, December 1982', *Journal of Applied Ecology*, 20(1), pp. 1-17. doi: 10.2307/2403372

Brand, J.M., Blum, M.S., Fales, H.M. and Pasteels, J.M. (1973) 'The chemistry of the defensive secretion of the beetle, *Drusilla canaliculata*', *Journal of Insect Physiology*, 19(2), pp. 369-382. doi: 10.1016/0022-1910(73)90112-1

Brändle, M., Durka, W. and Altmoos, M. (2000) 'Diversity of surface dwelling beetle assemblages in open-cast lignite mines in Central Germany', *Biodiversity and Conservation*, 9(9), pp. 1297-1311. doi: 10.1023/A:1008904605567

Brendell, M.J.D. (1975) *Coleoptera: Tenebrionidae.* London: Royal Entomological Society of London. Handbooks for the Identification of British Insects Vol. 5, Part 10.

Brenneisen, S. (2006) 'Space for urban wildlife: Designing green roofs as habitats in Switzerland', *Urban Habitats,* 4(1), pp. 27-36.

Britton, A.J., Marrs, R.H., Carey, P.D. and Pakeman, R.J. (2000) 'Comparison of techniques to increase *Calluna vulgaris* cover on heathland invaded by grasses in Breckland, south east England', *Biological Conservation*, 95(3), pp. 227-232. doi: 10.1016/S0006-3207(00)00047-1

Brook, A.J., Woodcock, B.A., Sinka, M. and Vanbergen, A.J. (2008) 'Experimental verification of suction sampler capture efficiency in grasslands of differing vegetation height and structure', *Journal of Applied Ecology*, 45(5), pp. 1357-1363. doi: 10.1111/j.1365-2664.2008.01530.x

Brooks, D.R., Bater, J.E., Clark, S.J., Monteith, D.T., Andrews, C., Corbett, S.J., Beaumont, D.A. and Chapman, J.W. (2012) 'Large carabid beetle declines in a United Kingdom monitoring network increases evidence for a widespread loss in insect biodiversity', *Journal of Applied Ecology*, 49(5), pp. 1009-1019. doi: 10.1111/j.1365-2664.2012.02194.x

Bröring, U., Mrzljak, J., Niedringhaus, R. and Wiegleb, G. (2005) 'Soil zoology I: arthropod communities in open landscapes of former brown coal mining areas', *Ecological Engineering*, 24(1–2), pp. 121-133. doi: 10.1016/j.ecoleng.2004.12.012

Bröring, U. and Wiegleb, G. (2005) 'Soil zoology II: colonization, distribution, and abundance of terrestrial Heteroptera in open landscapes of former brown coal mining areas', *Ecological Engineering*, 24(1), pp. 135-147. doi: 10.1016/j.ecoleng.2004.12.015

Brose, U. (2003a) 'Bottom-up control of carabid beetle communities in early successional wetlands: mediated by vegetation structure or plant diversity?', *Oecologia*, 135(3), pp. 407-13. doi: 10.1007/s00442-003-1222-7

Brose, U. (2003b) 'Island biogeography of temporary wetland carabid beetle communities', *Journal of Biogeography*, 30(6), pp. 879-888. doi: 10.1046/j.1365-2699.2003.00893.x

Brown, J.H. and Kodric-Brown, A. (1977) 'Turnover rates in insular biogeography: effect of immigration on extinction', *Ecology*, 58(2), pp. 445-449. doi: 10.2307/1935620

Bruland, G.L. and Richardson, C.J. (2005) 'Hydrologic, edaphic, and vegetative responses to microtopographic reestablishment in a restored wetland', *Restoration Ecology*, 13(3), pp. 515-523. doi: 10.1111/j.1526-100X.2005.00064.x

Brun, M., Di Pietro, F. and Bonthoux, S. (2017) 'Residents' perceptions and valuations of urban wastelands are influenced by vegetation structure'. To be published in *Urban Forestry & Urban Greening* [Preprint]. doi: 10.1016/j.ufug.2017.01.005.

Buchholz, S. (2009) 'Spider assemblages in an inland dune complex of Northwest Germany', *Drosera,* 2008(1/2), pp. 63-76.

(2010) 'Ground spider assemblages as indicators for habitat structure in inland sand ecosystems', *Biodiversity and Conservation*, 19(9), pp. 2565-2595. doi: 10.1007/s10531-010-9860-7

Buchholz, S. and Hannig, K. (2009) 'Do covers influence the capture efficiency of pitfall traps?', *European Journal of Entomology*, 106(4), pp. 667-671. doi: 10.14411/eje.2009.083

Buchholz, S., Hannig, K. and Schirmel, J. (2013) 'Losing uniqueness – shifts in carabid species composition during dry grassland and heathland succession', *Animal Conservation*, 16(6), pp. 661-670. doi: 10.1111/acv.12046

Buchholz, S., Jess, A.-M., Hertenstein, F. and Schirmel, J. (2010) 'Effect of the colour of pitfall traps on their capture efficiency of carabid beetles (Coleoptera: Carabidae), spiders (Araneae) and other arthropods', *European Journal of Entomology*, 107(2), pp. 277-280. doi: 10.14411/eje.2010.036

Buck, F.D. (1954) *Coleoptera: Lagriidae, Alleculidae, Tetratomidae, Melandryidae, Salpingidae, Pythidae, Mycteridae, Oedemeridae, Mordellidae, Scraptiidae, Pyrochroidae, Rhipiphoridae, Anthicidae, Aderidae and Meloidae.* London: Royal Entomological Society of London. Handbooks for the Identification of British Insects Vol. 5, Part 9.

Butt, K.R. and Briones, M.J.I. (2017) 'Earthworms and mesofauna from an isolated, alkaline chemical waste site in Northwest England', *European Journal of Soil Biology*, 78, pp. 43-49. doi: 10.1016/j.ejsobi.2016.11.005

Butterfly Conservation (2015) *Sloe carpet (Aleucis distinctata) factsheet*. Available at: http://butterfly-conservation.org/files/1.sloe-carpet-species-factsheet.pdf (Accessed: 5 October 2017).

Cameron, K.H. and Leather, S.R. (2012a) 'Heathland management effects on carabid beetle communities: the relationship between bare ground patch size and carabid biodiversity', *Journal of Insect Conservation*, 16(4), pp. 523-535. doi: 10.1007/s10841-011-9438-z

Cameron, K.H. and Leather, S.R. (2012b) 'How good are carabid beetles (Coleoptera, Carabidae) as indicators of invertebrate abundance and order richness?', *Biodiversity and Conservation*, 21(3), pp. 763-779. doi: 10.1007/s10531-011-0215-9

Cane, J.H. (1992) 'Soils of ground-nesting bees (Hymenoptera: Apoidea): texture, moisture, cell depth and climate', Journal of the Kansas Entomological Society, 64(4), pp. 406-413. (Accessed: 8 October 2017).

Cannings, R.A. (1989) 'The robber flies (Diptera: Asilidae) of a Festuca grassland in the Okanagan Valley, British Columbia', *Journal of the Entomological Society of British Columbia,* 86, pp. 14-26.

Capilla, X., Schwartz, C., Bedell, J.-P., Sterckeman, T., Perrodin, Y. and Morel, J.-L. (2006) 'Physicochemical and biological characterisation of different dredged sediment deposit sites in France', *Environmental Pollution*, 143(1), pp. 106-116. doi: 10.1016/j.envpol.2005.11.007

Carvell, C., Isaac, N., Jitlal, M., Peyton, J., Powney, G., Roy, D., Vanbergen, A., O'Connor, R., Jones, C. and Kunin, B. (2016) *Design and testing of a national pollinator and pollination monitoring framework*. Available at: http://randd.defra.gov.uk/Document.aspx?Document=13755_WC1101Finalreport.pdf (Accessed: 9 October 2017).

Carvell, C., Meek, W.R., Pywell, R.F., Goulson, D. and Nowakowski, M. (2007) 'Comparing the efficacy of agri-environment schemes to enhance bumble bee abundance and diversity on arable field margins', *Journal of Applied Ecology*, 44(1), pp. 29-40. doi: 10.1111/j.1365-2664.2006.01249.x

Casanova, M.T. and Brock, M.A. (2000) 'How do depth, duration and frequency of flooding influence the establishment of wetland plant communities?', *Plant Ecology*, 147(2), pp. 237-250. doi: 10.1023/A:1009875226637

Castleton, H.F., Stovin, V., Beck, S.B.M. and Davison, J.B. (2010) 'Green roofs; building energy savings and the potential for retrofit', *Energy and Buildings*, 42(10), pp. 1582-1591. doi: 10.1016/j.enbuild.2010.05.004

Catenazzi, A. and Donnelly, M.A. (2007) 'Role of supratidal invertebrates in the decomposition of beach-cast green algae *Ulva* sp.', *Marine Ecology Progress Series*, 349, pp. 33-42. doi: 10.3354/meps07106

Catherine Bickmore Associates (2004) *Northwick Road, Canvey Island: invertebrate survey 2002*. Unpublished report.

Chao, A., Gotelli, N.J., Hsieh, T.C., Sander, E.L., Ma, K.H., Colwell, R.K. and Ellison, A.M. (2014) 'Rarefaction and extrapolation with Hill numbers: a framework for sampling and estimation in species diversity studies', *Ecological Monographs*, 84(1), pp. 45-67. doi: 10.1890/13-0133.1

Chapin, F.S., Walker, L.R., Fastie, C.L. and Sharman, L.C. (1994) 'Mechanisms of primary succession following deglaciation at Glacier Bay, Alaska', *Ecological Monographs*, 64(2), pp. 149-175. doi: 10.2307/2937039

Chazdon, R.L., Careaga, S., Webb, C. and Vargas, O. (2003) 'Community and phylogenetic structure of reproductive traits of woody species in wet tropical forests', *Ecological Monographs*, 73(3), pp. 331-348. doi: 10.1890/02-4037

Chell, E., Uglow, J. and Howard-Birt, D. (2013) *Eclipse.* Challock, Kent: Stour Valley Arts.

Chilcutt, C.F. and Cowan, D.P. (1993) 'Methods for artificial rearing of solitary eumenid wasps (Hymenoptera: Vespidae: Eumeninae)', *Great Lakes Entomologist,* 26(1), pp. 15-19.

Churchill, T.B. and Arthur, J.M. (1999) 'Measuring spider richness: effects of different sampling methods and spatial and temporal scales', *Journal of Insect Conservation*, 3(4), pp. 287-295. doi: 10.1023/A:1009638706789

Cizek, L., Hauck, D. and Pokluda, P. (2012) 'Contrasting needs of grassland dwellers: habitat preferences of endangered steppe beetles (Coleoptera)', *Journal of Insect Conservation*, 16(2), pp. 281-293. doi: 10.1007/s10841-011-9415-6

Clarke, K.R. (1993) 'Non-parametric multivariate analyses of changes in community structure', *Austral Ecology*, 18(1), pp. 117-143. doi: 10.1111/j.1442-9993.1993.tb00438.x

Clarke, R.O.S. (1973) *Coleoptera: Heteroceridae.* London: Royal Entomological Society of London. Handbooks for the Identification of British Insects Vol. 5, Part 2(c).

Cleverly, J.F. (2002) *The impact of military disturbance on the chalk grassland ground beetle (Carabidae) fauna of Salisbury Plain Training Area.* Unpublished MSc thesis. University of Reading.

Cole, L.J., Pollock, M.L., Robertson, D., Holland, J.P., McCracken, D.I. and Harrison, W. (2010) 'The influence of fine-scale habitat heterogeneity on invertebrate assemblage structure in upland semi-natural grassland', *Agriculture, Ecosystems & Environment*, 136(1), pp. 69-80. doi: https://doi.org/10.1016/j.agee.2009.11.010

Colla, S.R., Willis, E. and Packer, L. (2009) 'Can green roofs provide habitat for urban bees (Hymenoptera: Apidae)?', *Cities and the Environment*, 2(1), 4 (12pp). doi: 10.15365/cate.2142009

Collinge, S.K. (2000) 'Effects of grassland fragmentation on insect species loss, colonization, and movement patterns', *Ecology*, 81(8), pp. 2211-2226. doi: 10.1890/0012-9658(2000)081[2211:EOGFOI]2.0.CO;2

Connell, J.H. (1983) 'On the prevalence and relative importance of interspecific competition: evidence from field experiments', *The American Naturalist*, 122(5), pp. 661-696. doi: 10.1086/284165

Connop, S. (2007) *Habitat and habitat management requirements of the shrill carder bee (Bombus sylvarum) and the brown-banded carder bee (Bombus humilis) in South Essex.* PhD thesis. University of East London. Available at: http://ethos.bl.uk/OrderDetails.do?uin=uk.bl.ethos.532654 (Accessed: 8 October 2017).

Connop, S. (2012) 'The beetle bump: innovative urban habitat creation for rare insects', *Essex Naturalist,* 29, pp. 108-113.

Connop, S., Hill, T., Steer, J. and Shaw, P. (2010) 'The role of dietary breadth in national bumblebee (*Bombus*) declines: simple correlation?', *Biological Conservation*, 143(11), pp. 2739-2746. doi: 10.1016/j.biocon.2010.07.021

Connop, S., Hill, T., Steer, J. and Shaw, P. (2011) 'Microsatellite analysis reveals the spatial dynamics of *Bombus humilis* and *Bombus sylvarum*', *Insect Conservation and Diversity*, 4(3), pp. 212-221. doi: 10.1111/j.1752-4598.2010.00116.x

Connop, S., Lindsay, R., Freeman, J., Clough, J., Kadas, G. and Nash, C. (2014) *TURAS multidisciplinary urban landscape design guidance: design, incorporation and monitoring of Barking Riverside brownfield landscaping*. Available at: http://www.turas-cities.org/uploads/milestone/file/11/TURAS_multidisciplinary_urban_landscape_design_guidance_Final1.pdf (Accessed: 9 October 2017).

Connop, S., Vandergert, P., Eisenberg, B., Collier, M.J., Nash, C., Clough, J. and Newport, D. (2016) 'Renaturing cities using a regionally-focused biodiversity-led multifunctional benefits approach to urban green infrastructure', *Environmental Science & Policy*, 62, pp. 99-111. doi: 10.1016/j.envsci.2016.01.013

Cook, R.D. and Weisberg, S. (1982) *Residuals and influence in regression.* London: Chapman and Hall.

Coombs, C.W. and Woodroffe, G.E. (1955) 'A revision of the British species of *Cryptophagus* (Herbst) (Coleoptera: Cryptophagidae)', *Ecological Entomology,* 106(6), pp. 237-264.

Cooter, J. (1996) 'Annotated keys to the British Leiodinae (Col., Leiodidae)', *Entomologist's Monthly Magazine,* 132, pp. 205-272.

Cooter, J. and Barclay, M.V.L. (2006) *A coleopterist's handbook.* 4th edn. Orpington: The Amateur Entomologists' Society.

Cornelisse, T.M., Vasey, M.C., Holl, K.D. and Letourneau, D.K. (2013) 'Artificial bare patches increase habitat for the endangered Ohlone tiger beetle (*Cicindela ohlone*)', *Journal of Insect Conservation*, 17(1), pp. 17-22. doi: 10.1007/s10841-012-9482-3

Cornelissen, J.H.C., Lavorel, S., Garnier, E., Díaz, S., Buchmann, N., Gurvich, D.E., Reich, P.B., ter Steege, H., Morgan, H.D. and van der Heijden, M.G.A. (2003) 'A handbook of protocols for standardised and easy measurement of plant functional traits worldwide', *Australian Journal of Botany*, 51(4), pp. 335-380. doi: 10.1071/BT02124

Cornett, M.W., Reich, P.B. and Puettmann, K.J. (1997) 'Canopy feedbacks and microtopography regulate conifer seedling distribution in two Minnesota conifer-deciduous forests', *Ecoscience*, 4(3), pp. 353-364. doi: 10.1080/11956860.1997.11682414

Cortet, J., Vauflery, A.G.-D., Poinsot-Balaguer, N., Gomot, L., Texier, C. and Cluzeau, D. (1999) 'The use of invertebrate soil fauna in monitoring pollutant effects', *European Journal of Soil Biology*, 35(3), pp. 115-134. doi: 10.1016/S1164-5563(00)00116-3

Corti, R., Larned, S.T. and Datry, T. (2013) 'A comparison of pitfall-trap and quadrat methods for sampling ground-dwelling invertebrates in dry riverbeds', *Hydrobiologia*, 717(1), pp. 13-26. doi: 10.1007/s10750-013-1563-0

Coudrain, V., Schüepp, C., Herzog, F., Albrecht, M. and Entling, M.H. (2014) 'Habitat amount modulates the effect of patch isolation on host-parasitoid interactions', *Frontiers in Environmental Science*, 2, (27pp). doi: 10.3389/fenvs.2014.00027

CPIE Chaîne des terrils (2014) *Contribuer à l'offre touristique du Bassin minier*. Available at: http://www.chainedesterrils.eu/indexcontenu_ssrub.php?srub=27&&rub=2 (Accessed: 5 Oct 2017.

Cracknell, B.E. (1968) *Portrait of London river: the tidal Thames from Teddington to the sea.* London: Robert Hale.

Cristofoli, S., Mahy, G., Kekenbosch, R. and Lambeets, K. (2010) 'Spider communities as evaluation tools for wet heathland restoration', *Ecological Indicators*, 10(3), pp. 773-780. doi: 10.1016/j.ecolind.2009.11.013

Cumming, G. and Calin-Jageman, R. (2017) *Introduction to the new statistics.* New York: Routledge.

Curreli, A., Wallace, H., Freeman, C., Hollingham, M., Stratford, C., Johnson, H. and Jones, L. (2013) 'Eco-hydrological requirements of dune slack vegetation and the implications of climate change', *Science of the Total Environment*, 443, pp. 910-919. doi: 10.1016/j.scitotenv.2012.11.035

Curry, J.P. (1969) 'The qualitative and quantitative composition of the fauna of an old grassland site at Celbridge, Co. Kildare', *Soil Biology and Biochemistry*, 1(3), pp. 219-227. doi: 10.1016/0038-0717(69)90022-4

D'Antonio, C. and Meyerson, L.A. (2002) 'Exotic plant species as problems and solutions in ecological restoration: a synthesis', *Restoration Ecology*, 10(4), pp. 703-713. doi: 10.1046/j.1526-100X.2002.01051.x

Darvill, B., Ellis, J.S., Lye, G.C. and Goulson, D. (2006) 'Population structure and inbreeding in a rare and declining bumblebee, *Bombus muscorum* (Hymenoptera: Apidae)', *Molecular Ecology*, 15(3), pp. 601-611. doi: 10.1111/j.1365-294X.2006.02797.x

Davenport, S. (2014) Conversation with James McGill, 17 October.

Davies, A. and Waite, S. (1998) 'The persistence of calcareous grassland species in the soil seed bank under developing and established scrub', *Plant Ecology*, 136(1), pp. 27-39. doi: 10.1023/A:1009759227900

Davies, Z.G., Edmondson, J.L., Heinemeyer, A., Leake, J.R. and Gaston, K.J. (2011) 'Mapping an urban ecosystem service: quantifying above-ground carbon storage at a city-wide scale', *Journal of Applied Ecology*, 48(5), pp. 1125-1134. doi: 10.1111/j.1365-2664.2011.02021.x

Davis, J. and Russell, L. (2015) 'The transport and beneficial re-use of Crossrail excavated material', in Black, M., Dodge, C. & Yu, J. (eds.) *Crossrail Project: infrastructure design and construction: Volume 2*. London: ICE Publishing, pp. 241-250.

Day, M.C. (1988) *Spider wasps: Hymenoptera: Pompilidae.* London: Royal Entomological Society of London. Handbooks for the Identification of British Insects Vol. 6, Part 4.

de Bello, F., Lepš, J. and Sebastià, M.T. (2005) 'Predictive value of plant traits to grazing along a climatic gradient in the Mediterranean', *Journal of Applied Ecology*, 42(5), pp. 824-833. doi: 10.1111/j.1365-2664.2005.01079.x

De Cáceres, M. (2013) *How to use the indicspecies package (ver. 1.7.1)*. Available at: https://cran.r-project.org/web/packages/indicspecies/vignettes/indicspeciesTutorial.pdf (Accessed: 9 October 2017).

De Cáceres, M. and Legendre, P. (2009) Indicspecies: associations between species and groups of sites: indices and statistical inference (Version 1.7.6) [R package]. Available at: https://cran.r-project.org/web/packages/indicspecies/index.html (Accessed: 7 October 2017).

de Keer, R. and Maelfait, J.-P. (1988) 'Observations on the life cycle of *Erigone atra* (Araneae, Erigoninae) in a heavily grazed pasture', *Pedobiologia,* 32, pp. 201-212.

De Marzo, L. (2016) 'Appunti di storia naturale sul coleottero ripicolo *Carpelimus obesus* (Kiesenwetter) (Staphylinidae Oxytelinae)', *Entomologica*, 35, pp. 77-87. doi: 10.15162/0425-1016/732

de Vries, H.H., den Boer, P.J. and van Dijk, T.S. (1996) 'Ground beetle species in heathland fragments in relation to survival, dispersal, and habitat preference', *Oecologia*, 107(3), pp. 332-342. doi: 10.1007/BF00328449

Debinski, D.M. and Holt, R.D. (2000) 'A survey and overview of habitat fragmentation experiments', *Conservation Biology*, 14(2), pp. 342-355. doi: 10.1046/j.1523-1739.2000.98081.x

den Boer, P.J. (1990) 'The survival value of dispersal in terrestrial arthropods', *Biological Conservation*, 54(3), pp. 175-192. doi: 10.1016/0006-3207(90)90050-Y

Dennis, P., Skartveit, J., Kunaver, A. and McCracken, D.I. (2015) 'The response of spider (Araneae) assemblages to structural heterogeneity and prey abundance in sub-montane vegetation modified by conservation grazing', *Global Ecology and Conservation*, 3, pp. 715-728. doi: 10.1016/j.gecco.2015.03.007

Dennis, P., Young, M.R. and Bentley, C. (2001) 'The effects of varied grazing management on epigeal spiders, harvestmen and pseudoscorpions of *Nardus stricta* grassland in upland Scotland', *Agriculture, Ecosystems & Environment*, 86(1), pp. 39-57. doi: 10.1016/S0167-8809(00)00263-2

Dennis, R.L.H., Dapporto, L., Dover, J.W. and Shreeve, T.G. (2013) 'Corridors and barriers in biodiversity conservation: a novel resource-based habitat perspective for butterflies', *Biodiversity and Conservation*, 22(12), pp. 2709-2734. doi: 10.1007/s10531-013-0540-2

Denton, J. (2002) 'Rare and uncommon Coleoptera records, 2001', *The Coleopterist,* 11(1), pp. 29-30.

Denys, C. and Schmidt, H. (1998) 'Insect communities on experimental mugwort (*Artemisia vulgaris* L.) plots along an urban gradient', *Oecologia*, 113(2), pp. 269-277. doi: 10.1007/s004420050378

Department for Communities and Local Government (England) (2006) *The Thames Gateway Interim Plan: Policy Framework*. Available at: http://webarchive.nationalarchives.gov.uk/20080306071505/http://www.communities.gov.uk/documents/thamesgateway/pdf/153605 (Accessed: 9 October 2017).

(England) (2007) *Thames Gateway: the Delivery Plan*. Available at: http://webarchive.nationalarchives.gov.uk/20100407230456/http://www.communities.gov.uk/documents/thamesgateway/pdf/565039.pdf (Accessed: 9 October 2017).

(England) (2015) *Building more homes on brownfield land: consultation proposals*. Available at: https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/398745/Brownfield_Consultation_Paper.pdf (Accessed: 9 October 2017).

Department for Environment, Food and Rural Affairs (England) (2011) *Biodiversity 2020: a strategy for England's wildlife and ecosystem services*. Available at: https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/69446/pb13583-biodiversity-strategy-2020-111111.pdf (Accessed: 9 October 2017).

Department of Environment (United Kingdom) (1994) *Biodiversity: the UK Action Plan* (Cm. 2428). Available at: http://jncc.defra.gov.uk/PDF/UKBAP_Action-Plan-1994.pdf (Accessed: 9 October 2017).

(United Kingdom) (1995) *The Thames Gateway Planning Framework: RPG 9a*. Available at: http://www.communities.gov.uk/documents/thamesgateway/pdf/147561.pdf (Accessed: 9 October 2017).

Department of the Environment, Transport and the Regions (England) (2000) *Planning Policy Guidance 3: Housing* (Cm. 4667). Available at: http://regulations.completepicture.co.uk/pdf/Planning/Planning%20Policy%20Guidance%203-%20Housing.pdf (Accessed: 10 October 2017).

Derunkov, A.V. (2004) 'Staphylinidae (Coleoptera) of the Pripiat River floodplain, National Park ‘Pripiatsky’, Belarus', *Acta Zoologica Lituanica*, 14(4), pp. 14-22. doi: 10.1080/13921657.2004.10512595

Desender, K., Maelfait, J.-P. and Baert, L. (2007) 'Ground beetles as ‘early warning-indicators’ in restored salt marshes and dune slacks', in Isermann, M. & Kiehl, K. (eds.) *Restoration of coastal ecoystems: Coastline Reports 7*. Leiden: EUCC – The Coastal Union, pp. 25-39.

Desender, K.R.C. (1996) 'Diversity and dynamics of coastal dune carabids', Annales Zoologici Fennici, 33(1), pp. 65-75. Available at: http://www.jstor.org/stable/23735405 (Accessed: 8 October 2017).

Dias, P.C. (1996) 'Sources and sinks in population biology', *Trends in Ecology & Evolution*, 11(8), pp. 326-330. doi: https://doi.org/10.1016/0169-5347(96)10037-9

Digweed, S.C., Currie, C.R., Cárcamo, H.A. and Spence, J.R. (1995) 'Digging out the "digging-in effect" of pitfall traps: influences of depletion and disturbance on catches of ground beetles (Coleoptera: Carabidae)', *Pedobiologia,* 39, pp. 561-576.

Dinno, A. (2016) dunn.test: Dunn's test of multiple comparisons using rank sums (Version 1.3.2) [R package]. Available at: https://cran.r-project.org/web/packages/indicspecies/index.html (Accessed: 7 October 2017).

Doležalová, J., Vojar, J., Smolová, D., Solský, M. and Kopecký, O. (2012) 'Technical reclamation and spontaneous succession produce different water habitats: a case study from Czech post-mining sites', *Ecological Engineering*, 43, pp. 5-12. doi: 10.1016/j.ecoleng.2011.11.017

Dolman, P.M. and Sutherland, W.J. (1994) 'The use of soil disturbance in the management of Breckland grass heaths for nature conservation', *Journal of Environmental Management*, 41(2), pp. 123-140. doi: 10.1006/jema.1994.1038

Dolný, A. and Harabiš, F. (2012) 'Underground mining can contribute to freshwater biodiversity conservation: allogenic succession forms suitable habitats for dragonflies', *Biological Conservation*, 145(1), pp. 109-117. doi: 10.1016/j.biocon.2011.10.020

Dormann, C.F., McPherson, J.M., Araújo, M.B., Bivand, R., Bolliger, J., Carl, G., Davies, R.G., Hirzel, A., Jetz, W., Daniel Kissling, W., Kühn, I., Ohlemüller, R., Peres-Neto, P.R., Reineking, B., Schröder, B., M. Schurr, F. and Wilson, R. (2007) 'Methods to account for spatial autocorrelation in the analysis of species distributional data: a review', *Ecography*, 30(5), pp. 609-628. doi: 10.1111/j.2007.0906-7590.05171.x

Downie, I.S., Ribera, I., McCracken, D.I., Wilson, W.L., Foster, G.N., Waterhouse, A., Abernethy, V.J. and Murphy, K.J. (2000) 'Modelling populations of *Erigone atra* and *E. dentipalpis* (Araneae: Linyphiidae) across an agricultural gradient in Scotland', *Agriculture, Ecosystems & Environment*, 80(1), pp. 15-28. doi: 10.1016/S0167-8809(00)00130-4

Dray, S. and Legendre, P. (2008) 'Testing the species-traits environment relationships: the fourth-corner problem revisited', *Ecology*, 89(12), pp. 3400-3412. doi: 10.1890/08-0349.1

Duff, A.G. (2012) *Beetles of Britain and Ireland Volume 1: Sphaeriusidae to Silphidae.* West Runton, Norfolk: A.G. Duff (Publishing).

(2016) *Beetles of Britain and Ireland Volume 4: Cerambycidae to Curculionidae.* West Runton, Norfolk: A.G. Duff (Publishing).

Dufrene, M. and Legendre, P. (1997) 'Species assemblages and indicator species: the need for a flexible asymmetrical approach', *Ecological*

Monographs, 67(3), pp. 345-366. doi: 10.1890/0012-9615(1997)067[0345:SAAIST]2.0.CO;2

Dunn, O.J. (1964) 'Multiple comparisons using rank sums', *Technometrics*, 6(3), pp. 241-252. doi: 10.2307/1266041

Edwards, A.R., Mortimer, S.R., Lawson, C.S., Westbury, D.B., Harris, S.J., Woodcock, B.A. and Brown, V.K. (2007) 'Hay strewing, brush harvesting of seed and soil disturbance as tools for the enhancement of botanical diversity in grasslands', *Biological Conservation*, 134(3), pp. 372-382. doi: 10.1016/j.biocon.2006.08.025

Edwards, M. (1994) *National lowland heathlands programme: bare ground invertebrates report*. Unpublished report.

(1995) *Visits to sites with bare ground created under the 1994 lowland heathland programme*. Unpublished report.

(1996) *Report on the utilisation of bare ground by insects at Albury Heath and Puttenham Common, Surrey, during June and August 1996*. Unpublished report.

(1997) *Monitoring of bare ground for use by heathland insects, interim report*. Unpublished report.

(1998) *Monitoring of bare ground for use by heathland insects*. Unpublished report.

Eggers, B., Matern, A., Drees, C., Eggers, J., Haerdtle, W. and Assmann, T. (2010) 'Value of semi-open corridors for simultaneously connecting open and wooded habitats: a case study with ground beetles', *Conservation Biology*, 24(1), pp. 256-266. doi: 10.1111/j.1523-1739.2009.01295.x

Ellis, J.S., Knight, M.E., Darvill, B. and Goulson, D. (2006) 'Extremely low effective population sizes, genetic structuring and reduced genetic diversity in a threatened bumblebee species, *Bombus sylvarum* (Hymenoptera: Apidae)', *Molecular Ecology*, 15(14), pp. 4375-4386. doi: 10.1111/j.1365-294X.2006.03121.x

Ellison, R.A., Woods, M.A., Allen, D.J., Forster, A., Pharaoh, T.C. and King, C. (2004) *Geology of London: Special memoir for 1: 50000 geological sheets 256 (north London), 257 (Romford), 270 (south London), and 271 (Dartford) (England and Wales).* Nottingham: British Geological Survey.

Else, G.R. and Edwards, M. (1996) 'Observations on *Osmia inermis* (Zetterstedt) and *O. uncinata* Gerstacker (Hym., Apidae) in the central Scottish Highlands', *Entomologist's Monthly Magazine,* 132, pp. 291-298.

English Nature (1991) *West Thurrock Lagoon & Marshes SSSI: citation*. Available at: https://necmsi.esdm.co.uk/PDFsForWeb/Citation/1006132.pdf (Accessed: 9 October 2017).

(2005a) *Canvey Wick SSSI: citation*. Available at: https://necmsi.esdm.co.uk/PDFsForWeb/Citation/2000497.pdf (Accessed: 9 October 2017).

(2005b) *Canvey Wick: views about management*. Available at: https://necmsi.esdm.co.uk/PDFsForWeb/VAM/2000497.pdf (Accessed: 9 October 2017).

Entling, W., Schmidt-Entling, M.H., Bacher, S., Brandl, R. and Nentwig, W. (2010) 'Body size–climate relationships of European spiders', *Journal of Biogeography*, 37(3), pp. 477-485. doi: 10.1111/j.1365-2699.2009.02216.x

Entling, W., Schmidt, M.H., Bacher, S., Brandl, R. and Nentwig, W. (2007) 'Niche properties of Central European spiders: shading, moisture and the evolution of the habitat niche', *Global Ecology and Biogeography*, 16(4), pp. 440-448. doi: 10.1111/j.1466-8238.2006.00305.x

Evans, H.E. (1974) 'Digger wasps as colonizers of new habitat (Hymenoptera: Aculeata)', Journal of the New York Entomological Society, LXXXII, pp. 259-267. Available at: http://www.jstor.org/stable/25008941 (Accessed: 8 October 2017).

Evans, M.E.G. (1977) 'Locomotion in the Coleoptera Adephaga, especially Carabidae', *Journal of Zoology*, 181(2), pp. 189-226. doi: 10.1111/j.1469-7998.1977.tb03237.x

Evans, M.E.G. and Forsythe, T.G. (1984) 'A comparison of adaptations to running, pushing and burrowing in some adult Coleoptera: especially Carabidae', *Journal of Zoology*, 202(4), pp. 513-534. doi: 10.1111/j.1469-7998.1984.tb05049.x

Everaars, J., Strohbach, M.W., Gruber, B. and Dormann, C.F. (2011) 'Microsite conditions dominate habitat selection of the red mason bee (*Osmia bicornis*, Hymenoptera: Megachilidae) in an urban environment: A case study from Leipzig, Germany', *Landscape and Urban Planning*, 103(1), pp. 15-23. doi: 10.1016/j.landurbplan.2011.05.008

Eversham, B.C., Roy, D.B. and Telfer, M.G. (1996) 'Urban, industrial and other manmade sites as analogues of natural habitats for Carabidae', Annales Zoologici Fennici, 33(1), pp. 149-156. Available at: http://www.jstor.org/stable/23735414 (Accessed: 8 October 2017).

Ewald, J.A., Wheatley, C.J., Aebischer, N.J., Moreby, S.J., Duffield, S.J., Crick, H.Q.P. and Morecroft, M.B. (2015) 'Influences of extreme weather, climate and

pesticide use on invertebrates in cereal fields over 42 years', *Global Change Biology,* 21(11), pp. 3931-3950.

Ewing, K. (2002) 'Mounding as a technique for restoration of prairie on a capped landfill in the Puget Sound Lowlands', *Restoration Ecology*, 10(2), pp. 289-296. doi: 10.1046/j.1526-100X.2002.09976.x

Exeler, N., Kratochwil, A. and Hochkirch, A. (2009) 'Restoration of riverine inland sand dune complexes: implications for the conservation of wild bees', *Journal of Applied Ecology*, 46(5), pp. 1097-1105. doi: 10.1111/j.1365-2664.2009.01701.x

Eyre, M.D., Lott, D.A. and Luff, M.L. (2001) 'The rove beetles (Coleoptera, Staphylinidae) of exposed riverine sediments in Scotland and Northern England: habitat classification and conservation aspects', *Journal of Insect Conservation*, 5(3), pp. 173-186. doi: 10.1023/a:1017956927482

Eyre, M.D., Luff, M.L. and Woodward, J.C. (2003) 'Beetles (Coleoptera) on brownfield sites in England: an important conservation resource?', *Journal of Insect Conservation*, 7(4), pp. 223-231. doi: 10.1023/B:JICO.0000021020.66549.1e

Falk, S.J. (2015) *Field guide to the bees of Great Britain and Ireland.* London: British Wildlife Publishing.

Falk, S.J. and Pont, A.C. (1996) *A review of the scarce and threatened flies of Great Britain: Part X: Calyptrata*. Unpublished report.

Falk, S.J. and Pont, A.C. (2017) *A provisional assessment of the status of Calypterate flies in the UK*. Available at: http://publications.naturalengland.org.uk/publication/5145257026519040 (Accessed: 9 October 2017).

Farley, P. and Roberts, M.S. (2012) *Edgelands: journeys into England's true wilderness.* London: Random House.

Fattorini, S., Maltzeff, P. and Salvati, L. (2015) 'Use of insect distribution across landscape-soil units to assess conservation priorities in a Mediterranean coastal reserve: the tenebrionid beetles of Castelporziano (Central Italy)', *Rendiconti Lincei*, 26(3), pp. 353-366. doi: 10.1007/s12210-015-0391-8

Fawki, S. and Toft, S. (2005) 'Food preferences and the value of animal food for the carabid beetle *Amara similata* (Gyll.) (Col., Carabidae)', *Journal of Applied Entomology*, 129(9-10), pp. 551-556. doi: 10.1111/j.1439-0418.2005.00992.x

Field, C.D., Dise, N.B., Payne, R.J., Britton, A.J., Emmett, B.A., Helliwell, R.C., Hughes, S., Jones, L., Lees, S., Leake, J.R., Leith, I.D., Phoenix, G.K., Power, S.A., Sheppard, L.J., Southon, G.E., Stevens, C.J. and Caporn, S.J.M. (2014) 'The role of nitrogen deposition in widespread plant community change across

semi-natural habitats', *Ecosystems*, 17(5), pp. 864-877. doi: 10.1007/s10021-014-9765-5

Field, J. (1989) 'Intraspecific parasitism and nesting success in the solitary wasp Ammophila sabulosa', *Behaviour*, 110(1), pp. 23-45. doi: 10.1163/156853989X00367

Fischer, J. and Lindenmayer, D.B. (2007) 'Landscape modification and habitat fragmentation: a synthesis', *Global Ecology and Biogeography*, 16(3), pp. 265-280. doi: 10.1111/j.1466-8238.2007.00287.x

Fitton, M.G. (1973) *Studies on the biology and ecology of Cantharidae (Coleoptera).* PhD thesis. University of London. Available at: http://ethos.bl.uk/OrderDetails.do?uin=uk.bl.ethos.643991 (Accessed: 8 October 2017).

Florencio, M., Serrano, L., Gómez-Rodríguez, C., Millán, A. and Díaz-Paniagua, C. (2009) 'Inter- and intra-annual variations of macroinvertebrate assemblages are related to the hydroperiod in Mediterranean temporary ponds', *Hydrobiologia*, 634(1), pp. 167-183. doi: 10.1007/s10750-009-9897-3

Ford, H., Garbutt, A., Jones, L. and Jones, D.L. (2013) 'Grazing management in saltmarsh ecosystems drives invertebrate diversity, abundance and functional group structure', *Insect Conservation and Diversity*, 6(2), pp. 189-200. doi: 10.1111/j.1752-4598.2012.00202.x

Forsythe, T.G. (1981) 'Running and pushing in relationship to hind leg structure in some Carabidae (Coleoptera)', The Coleopterists Bulletin, 35(4), pp. 353-378. Available at: http://www.jstor.org/stable/4007954 (Accessed: 8 October 2017).

Fortel, L., Henry, M., Guilbaud, L., Mouret, H. and Vaissière, B.E. (2016) 'Use of human-made nesting structures by wild bees in an urban environment', *Journal of Insect Conservation*, 20(2), pp. 239-253. doi: 10.1007/s10841-016-9857-y

Foster, G.N., Bilton, D.T. and Friday, L.E. (2014) *Keys to adults of the water beetles of Britain and Ireland (Part 2): (Coleoptera: Polyphaga: Hydrophiloidea - both aquatic and terrestrial species).* St. Albans: Royal Entomological Society. Handbooks for the Identification of British Insects Vol. 4, Part 5(b).

Foster, G.N. and Friday, L.E. (2011) *Keys to adults of the water beetles of Britain and Ireland (Part 1): (Coleoptera: Hydradephaga: Gyrinidae, Haliplidae, Paelobiidae, Noteridae and Dytiscidae).* 2nd edn. St. Albans: Royal Entomological Society. Handbooks for the Identification of British Insects Vol. 4, Part 5.

Fountain-Jones, N.M., Baker, S.C. and Jordan, G.J. (2015) 'Moving beyond the guild concept: developing a practical functional trait framework for terrestrial beetles', *Ecological Entomology*, 40(1), pp. 1-13. doi: 10.1111/een.12158

Fournier, B., Gillet, F., Le Bayon, R.C., Mitchell, E.A. and Moretti, M. (2015) 'Functional responses of multitaxa communities to disturbance and stress gradients in a restored floodplain', *Journal of Applied Ecology*, 52(5), pp. 1364-1373. doi: 10.1111/1365-2664.12493

Fox, J. and Weisberg, S. (2010) *An R companion to applied regression.* 2nd edn. London: Sage Publications Ltd.

Freude, H., Harde, K.W. and Lohse, G.A. (1967) *Die Käfer Mitteleuropas: Band 7: Clavicornia.* Krefeld, Germany: Goecke & Evers Verlag.

(1974) *Die Käfer Mitteleuropas: Band 5: Staphylinidae II (Hypocyphtinae und Aleocharinae), Pselaphidae.* Krefeld, Germany: Goecke & Evers Verlag.

Fried, J.H., Levey, D.J. and Hogsette, J.A. (2005) 'Habitat corridors function as both drift fences and movement conduits for dispersing flies', *Oecologia*, 143(4), pp. 645-651. doi: 10.1007/s00442-005-0023-6

Fussell, M. and Corbet, S.A. (1992) 'The nesting places of some British bumble bees', *Journal of Apicultural Research*, 31(1), pp. 32-41. doi: 10.1080/00218839.1992.11101258

Gagic, V., Bartomeus, I., Jonsson, T., Taylor, A., Winqvist, C., Fischer, C., Slade, E.M., Steffan-Dewenter, I., Emmerson, M. and Potts, S.G. (2015) 'Functional identity and diversity of animals predict ecosystem functioning better than species-based indices', *Proceedings of the Royal Society of London B: Biological Sciences*, 282(1801), 20142620 (8pp). doi: 10.1098/rspb.2014.2620

García-Tejero, S. and Taboada, Á. (2016) 'Microhabitat heterogeneity promotes soil fertility and ground-dwelling arthropod diversity in Mediterranean wood-pastures', *Agriculture, Ecosystems & Environment*, 233, pp. 192-201. doi: 10.1016/j.agee.2016.09.004

García, R.R., Garcia, U., Osoro, K. and Celaya, R. (2011) 'Ground-dwelling arthropod assemblages of partially improved heathlands according to the species of grazer and grazing regime', *European Journal of Entomology*, 108(1), pp. 107-115. doi: 10.14411/eje.2011.014

Gathmann, A., Greiler, H.-J. and Tscharntke, T. (1994) 'Trap-nesting bees and wasps colonizing set-aside fields: succession and body size, management by cutting and sowing', *Oecologia*, 98(1), pp. 8-14. doi: 10.1007/BF00326084

Gathmann, A. and Tscharntke, T. (2002) 'Foraging ranges of solitary bees', *Journal of Animal Ecology*, 71(5), pp. 757-764. doi: 10.1046/j.1365-2656.2002.00641.x

Gerisch, M. (2014) 'Non-random patterns of functional redundancy revealed in ground beetle communities facing an extreme flood event', *Functional Ecology*, 28(6), pp. 1504-1512. doi: 10.1111/1365-2435.12272

Gerisch, M., Agostinelli, V., Henle, K. and Dziock, F. (2012) 'More species, but all do the same: contrasting effects of flood disturbance on ground beetle functional and species diversity', *Oikos*, 121(4), pp. 508-515. doi: 10.1111/j.1600-0706.2011.19749.x

Gerisch, M., Schanowski, A., Figura, W., Gerken, B., Dziock, F. and Henle, K. (2006) 'Carabid beetles (Coleoptera, Carabidae) as indicators of hydrological site conditions in floodplain grasslands', *International Review of Hydrobiology*, 91(4), pp. 326-340. doi: 10.1002/iroh.200610888

Gerken, B., Dörfer, K., Buschmann, M., Kamps-Schwob, S., Berthelmann, J. and Gertenbach, D. (1991) 'Composition and distribution of carabid communities along rivers and ponds in the region of Upper Weser (NW/NDS/FRG) with respect to protection and management of a floodplain ecosystem', *Regulated Rivers: Research & Management*, 6(4), pp. 313-320. doi: 10.1002/rrr.3450060409

Gerlach, A., Voigtländer, K. and Heidger, C.M. (2009) 'Influences of the behaviour of epigeic arthropods (Diplopoda, Chilopoda, Carabidae) on the efficiency of pitfall trapping', *Soil Organisms,* 81(3), pp. 773-790.

Ghazoul, J. (2001) 'Effect of soil hardness on aggression in the solitary wasp *Mellinus arvensis*', *Ecological Entomology*, 26(5), pp. 457-466. doi: 10.1046/j.1365-2311.2001.00348.x

Gibson, C.W.D. (1998) *Brownfield: red data: the values artificial habitats have for uncommon invertebrates*. Available at: http://publications.naturalengland.org.uk/publication/127046 (Accessed: 9 October 2017).

Gibson, C.W.D., Brown, V.K., Losito, L. and McGavin, G.C. (1992) 'The response of invertebrate assemblies to grazing', *Ecography*, 15(2), pp. 166-176. doi: 10.1111/j.1600-0587.1992.tb00020.x

Gilbert, O.L. (1989) *The ecology of urban habitats.* London: Chapman and Hall.

Gilhaus, K., Vogt, V. and Hölzel, N. (2015) 'Restoration of sand grasslands by topsoil removal and self-greening', *Applied Vegetation Science*, 18(4), pp. 661-673. doi: 10.1111/avsc.12176

Gobbi, M. and Fontaneto, D. (2008) 'Biodiversity of ground beetles (Coleoptera: Carabidae) in different habitats of the Italian Po lowland', *Agriculture, Ecosystems & Environment*, 127(3), pp. 273-276. doi: 10.1016/j.agee.2008.04.011

Godefroid, S., Sansen, U. and Koedam, N. (2016) 'Long-term influence of sod cutting depth on the restoration of degraded wet heaths', *Restoration Ecology*, 25(2), pp. 191-200. doi: 10.1111/rec.12412

Good, J.A. and Giller, P.S. (1991) 'The effect of cereal and grass management on staphylinid (Coleoptera) assemblages in south-west Ireland', *Journal of Applied Ecology*, 28(3), pp. 810-826. doi: 10.2307/2404209

Goodwin, C., Keep, B. and Leather, S.R. (2017) 'Habitat richness and tree species richness of roundabouts: effects on site selection and the prevalence of arboreal caterpillars', *Urban Ecosystems,* 20(4), pp. 889-895.

Gossner, M.M., Simons, N.K., Achtziger, R., Blick, T., Dorow, W.H.O., Dziock, F., Köhler, F., Rabitsch, W. and Weisser, W.W. (2015) 'A summary of eight traits of Coleoptera, Hemiptera, Orthoptera and Araneae, occurring in grasslands in Germany', *Scientific Data*, 2, 150013 (9pp). doi: 10.1038/sdata.2015.13

Gough, M.W. and Marrs, R.H. (1990) 'A comparison of soil fertility between semi-natural and agricultural plant communities: implications for the creations of species-rich grassland on abandoned agricultural land', *Biological Conservation*, 51(2), pp. 83-96. doi: 10.1016/0006-3207(90)90104-W

Goulson, D., Nicholls, E., Botías, C. and Rotheray, E.L. (2015) 'Bee declines driven by combined stress from parasites, pesticides, and lack of flowers', *Science*, 347(6229), 1255957 (9pp). doi: 10.1126/science.1255957

Gower, J.C. (1971) 'A general coefficient of similarity and some of its properties', *Biometrics*, 27(4), pp. 857-871. doi: 10.2307/2528823

Greater London Authority (England) (2008) *Living roofs and walls technical report: supporting London Plan Policy* Greater London Authority. Available at: https://www.london.gov.uk/sites/default/files/living-roofs.pdf (Accessed: 9 October 2017).

(England) (2016) *The London Plan: the spatial development strategy for London consolidated with alterations since 2011*. Available at: https://www.london.gov.uk/sites/default/files/the_london_plan_malp_final_for_web_0606_0.pdf (Accessed: 9 October 2017).

Greenslade, P.J.M. (1964) 'Pitfall trapping as a method for studying populations of Carabidae (Coleoptera)', *Journal of Animal Ecology*, 33(2), pp. 301-310. doi: 10.2307/2632

Greenwood, B. and Gemmell, R.P. (1978) 'Derelict industrial land as a habitat for rare plants in S. Lancs. (v.c. 59) and W. Lancs. (v. c. 60)', *Watsonia,* 12, pp. 33-40.

Gregor, F., Rozkošný, R., Barták, M. and Vaňhara, J. (2016) *Manual of Central European Muscidae (Diptera): morphology, taxonomy, identification and distribution.* Stuttgart: Schweizerbart Science Publishers.

Gregory, W.W. and Musick, G.J. (1976) 'Insect management in reduced tillage systems', *Bulletin of the Entomological Society of America*, 22(3), pp. 302-304. doi: 10.1093/besa/22.3.302

Grootjans, A.P., Dullo, B.S., Kooijman, A.M., Bekker, R.M. and Aggenbach, C. (2013) 'Restoration of dune vegetation in the Netherlands', in Martínez, M.L., Gallego-Fernández, J.B. & Hesp, P.A. (eds.) *Restoration of coastal dunes*. Heidelberg: Springer, pp. 235-253.

Grundel, R., Jean, R.P., Frohnapple, K.J., Glowacki, G.A., Scott, P.E. and Pavlovic, N.B. (2010) 'Floral and nesting resources, habitat structure, and fire influence bee distribution across an open-forest gradient', *Ecological Applications*, 20(6), pp. 1678-1692. doi: 10.1890/08-1792.1

Günther, J. and Assmann, T. (2005) 'Restoration ecology meets carabidology: effects of floodplain restitution on ground beetles (Coleoptera, Carabidae)', *Biodiversity and Conservation*, 14(7), pp. 1583-1606. doi: 10.1007/s10531-004-0531-4

Gutiérrez, D., León-Cortés, J.L., Menéndez, R., Wilson, R.J., Cowley, M.J. and Thomas, C.D. (2001) 'Metapopulations of four lepidopteran herbivores on a single host plant, *Lotus corniculatus*', *Ecology*, 82(5), pp. 1371-1386. doi: 10.1890/0012-9658(2001)082[1371:MOFLHO]2.0.CO;2

Habel, J.C., Husemann, M., Schmitt, T., Dapporto, L., Rödder, D. and Vandewoestijne, S. (2013) 'A forest butterfly in Sahara Desert oases: isolation does not matter', *Journal of Heredity*, 104(2), pp. 234-247. doi: 10.1093/jhered/ess092

Haddad, N.M. (1999) 'Corridor and distance effects on interpatch movements: a landscape experiment with butterflies', *Ecological Applications*, 9(2), pp. 612-622. doi: 10.1890/1051-0761(1999)009[0612:CADEOI]2.0.CO;2

Haddad, N.M., Bowne, D.R., Cunningham, A., Danielson, B.J., Levey, D.J., Sargent, S. and Spira, T. (2003) 'Corridor use by diverse taxa', *Ecology*, 84(3), pp. 609-615. doi: 10.1890/0012-9658(2003)084[0609:CUBDT]2.0.CO;2

Haddad, N.M. and Tewksbury, J.J. (2005) 'Low-quality habitat corridors as movement conduits for two butterfly species', *Ecological Applications*, 15(1), pp. 250-257. doi: 10.1890/03-5327

Hall, I.G. (1957) 'The ecology of disused pit heaps in England', *Journal of Ecology*, 45(3), pp. 689-720. doi: 10.2307/2256953

Hallmann, C.A., Sorg, M., Jongejans, E., Siepel, H., Hofland, N., Schwan, H., Stenmans, W., Müller, A., Sumser, H., Hörren, T., Goulson, D. and de Kroon, H. (2017) 'More than 75 percent decline over 27 years in total flying insect biomass in protected areas', *PLOS ONE,* 12(10), p. e0185809.

Hambleton, J.P. and Drescher, A. (2009) 'Modeling wheel-induced rutting in soils: rolling', *Journal of Terramechanics*, 46(2), pp. 35-47. doi: 10.1016/j.jterra.2009.02.003

Hammond, P.M. (1972) 'Notes on British Staphylinidae 3. The British species of *Sepedophilus* Gistel (*Conosomus* auctt.)', *Entomologist's Monthly Magazine,* 108, pp. 130-165.

(1994) 'Practical approaches to the estimation of the extent of biodiversity in speciose groups', *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, 345(1311), pp. 119-136. doi: 10.1098/rstb.1994.0092

Hammond, P.M. and Harvey, P.R. (2008) '*Scybalicus oblongiusculus* (Dejean) (Carabidae) apparently thriving on brownfield sites in the East Thames Corridor', *The Coleopterist,* 17(1), pp. 13-17.

Hancock, M.H. and Legg, C.J. (2012) 'Pitfall trapping bias and arthropod body mass', *Insect Conservation and Diversity*, 5(4), pp. 312-318. doi: 10.1111/j.1752-4598.2011.00162.x

Harabiš, F. and Dolný, A. (2012) 'Human altered ecosystems: suitable habitats as well as ecological traps for dragonflies (Odonata): the matter of scale', *Journal of Insect Conservation*, 16(1), pp. 121-130. doi: 10.1007/s10841-011-9400-0

Harabiš, F., Tichanek, F. and Tropek, R. (2013) 'Dragonflies of freshwater pools in lignite spoil heaps: restoration management, habitat structure and conservation value', *Ecological Engineering*, 55, pp. 51-61. doi: 10.1016/j.ecoleng.2013.02.007

Härdtle, W., Niemeyer, M., Niemeyer, T., Assmann, T. and Fottner, S. (2006) 'Can management compensate for atmospheric nutrient deposition in heathland ecosystems?', *Journal of Applied Ecology*, 43(4), pp. 759-769. doi: 10.1111/j.1365-2664.2006.01195.x

Harrison, C. and Davies, G. (2002) 'Conserving biodiversity that matters: practitioners' perspectives on brownfield development and urban nature conservation in London', *Journal of Environmental Management*, 65(1), pp. 95-108. doi: 10.1006/jema.2002.0539

Harvey, J.A., van der Putten, W.H., Turin, H., Wagenaar, R. and Bezemer, T.M. (2008) 'Effects of changes in plant species richness and community traits on carabid assemblages and feeding guilds', *Agriculture, Ecosystems & Environment*, 127(1), pp. 100-106. doi: 10.1016/j.agee.2008.03.006

Harvey, P. (2000) 'The East Thames Corridor: a nationally important invertebrate fauna under threat', *British Wildlife,* 12(2), pp. 91-98.

(2001) 'An appraisal of the status and ecology of *Gymnosoma nitens* (Meigen) (Diptera: Tachinidae) in Britain and its relationship with the fauna of the Thames terrace grasslands of the East Thames corridor', *British Journal of Entomology and Natural History,* 14(4), pp. 225-231.

Harvey, P. (2006) 'The jumping spider *Sitticus distinguendus* (Simon, 1868), new to Britain and Essex', *Essex Naturalist,* 23, pp. 34-35.

Harvey, P. (2008) *Thames Gateway brownfields: invertebrate biodiversity and management*. Unpublished report.

Harvey, P. and Murphy, J. (1985) '*Zodarion italicum* (Araneae: Zodariidae), a species newly recorded in Britain', *Newsletter of the British Arachnological Society,* 44, p. 4.

Harvey, P.R. (2005) *Invertebrate survey and assessment for West Thurrock lagoons 2005*. Unpublished report.

(2007) 'The 2007 AGM Address. Brownfield invertebrates in Essex - nationally important and under threat', *Essex Naturalist,* 24, pp. 8-14.

(2012) *Canvey Wick invertebrate survey report*. Unpublished report.

Harvey, P.R., Nellist, D.R. and Telfer, M.G. (2002) *Provisional atlas of British spiders (Arachnida, Araneae), Volume 1.* Huntingdon: Biological Records Centre.

Haynes, R.J. (2009) 'Reclamation and revegetation of fly ash disposal sites – Challenges and research needs', *Journal of Environmental Management*, 90(1), pp. 43-53. doi: 10.1016/j.jenvman.2008.07.003

Helden, A.J., Stamp, G.C. and Leather, S.R. (2012) 'Urban biodiversity: comparison of insect assemblages on native and non-native trees', *Urban Ecosystems*, 15(3), pp. 611-624. doi: 10.1007/s11252-012-0231-x

Hemachandra, K.S., Holliday, N.J., Mason, P.G., Soroka, J.J. and Kuhlmann, U. (2007) 'Comparative assessment of the parasitoid community of *Delia radicum* in the Canadian prairies and Europe: a search for classical biological control agents', *Biological Control*, 43(1), pp. 85-94. doi: 10.1016/j.biocontrol.2007.07.005

Hendrychová, M., Šálek, M., Tajovský, K. and Řehoř, M. (2012) 'Soil properties and species richness of invertebrates on afforested sites after brown coal mining', *Restoration Ecology*, 20(5), pp. 561-567. doi: 10.1111/j.1526-100X.2011.00841.x

Heneberg, P. (2012) 'Flagship bird species habitat management supports the presence of ground-nesting aculeate hymenopterans', *Journal of Insect Conservation*, 16(6), pp. 899-908. doi: 10.1007/s10841-012-9477-0

Heneberg, P., Bogusch, P. and Řehounek, J. (2013) 'Sandpits provide critical refuge for bees and wasps (Hymenoptera: Apocrita)', *Journal of Insect Conservation*, 17(3), pp. 473-490. doi: 10.1007/s10841-012-9529-5

Heneberg, P., Bogusch, P. and Řezáč, M. (2016a) 'Off-road motorcycle circuits support long-term persistence of bees and wasps (Hymenoptera: Aculeata) of open landscape at newly formed refugia within otherwise afforested temperate landscape', *Ecological Engineering*, 93, pp. 187-198. doi: 10.1016/j.ecoleng.2016.05.026

Heneberg, P., Hesoun, P. and Skuhrovec, J. (2016b) 'Succession of arthropods on xerothermophilous habitats formed by sand quarrying: epigeic beetles (Coleoptera) and orthopteroids (Orthoptera, Dermaptera and Blattodea)', *Ecological Engineering*, 95, pp. 340-356. doi: 10.1016/j.ecoleng.2016.06.022

Heneberg, P. and Řezáč, M. (2014) 'Dry sandpits and gravel-sandpits serve as key refuges for endangered epigeic spiders (Araneae) and harvestmen (Opiliones) of Central European steppes aeolian sands', *Ecological Engineering*, 73, pp. 659-670. doi: 10.1016/j.ecoleng.2014.09.101

Herman, L.H. (1986) 'Revision of Bledius. Part IV. Classification of species groups, phylogeny, natural history, and catalogue (Coleoptera, Staphylinidae, Oxytelinae)', *Bulletin of the American Museum of Natural History,* 184(1), pp. 1-368.

Hill, M.O. and Gauch, H.G. (1980) 'Detrended Correspondence Analysis: an improved ordination technique', in van der Maarel, E. (ed.) *Classification and Ordination: Symposium on advances in vegetation science, Nijmegen, The Netherlands, May 1979*. Dordrecht: Springer Netherlands, pp. 47-58.

Hobbs, R.J. and Norton, D.A. (1996) 'Towards a conceptual framework for restoration ecology', *Restoration Ecology*, 4(2), pp. 93-110. doi: 10.1111/j.1526-100X.1996.tb00112.x

Hodecek, J., Kuras, T., Sipos, J. and Dolny, A. (2015) 'Post-industrial areas as successional habitats: long-term changes of functional diversity in beetle communities', *Basic and Applied Ecology*, 16(7), pp. 629-640. doi: 10.1016/j.baae.2015.06.004

Hodge, P.J. (1989) *A survey of the insects of the Somerset moors peat production zones with recommendations for their conservation*. Unpublished report.

(2011) '*Pseudoperapion brevirostre* (Herbst, 1797) (Apionidae) new to the British Isles', *The Coleopterist,* 20(3), pp. 111-115.

Hoffmann, H., Michalik, P., Görn, S. and Fischer, K. (2016) 'Effects of fen management and habitat parameters on staphylinid beetle (Coleoptera:

Staphylinidae) assemblages in north-eastern Germany', *Journal of Insect Conservation*, 20(1), pp. 129-139. doi: 10.1007/s10841-016-9847-0

Holec, M. and Frouz, J. (2005) 'Ant (Hymenoptera: Formicidae) communities in reclaimed and unreclaimed brown coal mining spoil dumps in the Czech Republic', *Pedobiologia*, 49(4), pp. 345-357. doi: 10.1016/j.pedobi.2005.03.001

Holm, S. (1979) 'A simple sequentially rejective multiple test procedure', *Scandinavian Journal of Statistics*, 6(2), pp. 65-70. doi: 10.2307/4615733

Homburg, K., Homburg, N., Schäfer, F., Schuldt, A. and Assmann, T. (2014) 'Carabids.org – a dynamic online database of ground beetle species traits (Coleoptera, Carabidae)', *Insect Conservation and Diversity*, 7(3), pp. 195-205. doi: 10.1111/icad.12045

Honek, A. (1988) 'The effect of crop density and microclimate on pitfall trap catches of Carabidae, Staphylinidae (Coleoptera), and Lycosidae (Araneae) in cereal fields', *Pedobiologia,* 32(3-4), pp. 233-242.

Honek, A., Martinkova, Z., Saska, P. and Koprdova, S. (2009) 'Role of post-dispersal seed and seedling predation in establishment of dandelion (*Taraxacum* agg.) plants', *Agriculture, Ecosystems & Environment*, 134(1), pp. 126-135. doi: 10.1016/j.agee.2009.06.001

Honnay, O., Verheyen, K., Butaye, J., Jacquemyn, H., Bossuyt, B. and Hermy, M. (2002) 'Possible effects of habitat fragmentation and climate change on the range of forest plant species', *Ecology Letters*, 5(4), pp. 525-530. doi: 10.1046/j.1461-0248.2002.00346.x

Hopkin, S.P. and Martin, M.H. (1985) 'Assimilation of zinc, cadmium, lead, copper, and iron by the spider *Dysdera crocata*, a predator of woodlice', *Bulletin of Environmental Contamination and Toxicology*, 34(1), pp. 183-187. doi: 10.1007/BF01609722

Hough-Snee, N., Long, A.L., Jeroue, L. and Ewing, K. (2011) 'Mounding alters environmental filters that drive plant community development in a novel grassland', *Ecological Engineering*, 37(11), pp. 1932-1936. doi: 10.1016/j.ecoleng.2011.06.013

Howe, M.A. (2015) 'Coastal soft cliff invertebrates are reliant upon dynamic coastal processes', *Journal of Coastal Conservation*, 19(6), pp. 809-820. doi: 10.1007/s11852-015-0374-z

Hsieh, T.C., Ma, K.H. and Chao, A. (2016) iNEXT: iNterpolation and EXTrapolation for species diversity (Version 2.0.12) [R package]. Available at: https://cran.r-project.org/web/packages/iNEXT/index.html (Accessed: 7 October 2017).

Hubble, D. (2012) *Keys to the adults of seed and leaf beetles of Britain and Ireland.* Telford: Field Studies Council.

Hurlbert, S.H. (1984) 'Pseudoreplication and the design of ecological field experiments', *Ecological Monographs*, 54(2), pp. 187-211. doi: 10.2307/1942661

Hutchings, M.J. and Booth, K.D. (1996) 'Studies on the feasibility of re-creating chalk grassland vegetation on ex-arable land. I. The potential roles of the seed bank and the seed rain', *Journal of Applied Ecology*, 33(5), pp. 1171-1181. doi: 10.2307/2404696

Hyman, P.S. and Parsons, M.S. (1994) *A review of the scarce and threatened Coleoptera of Great Britain: Part 2.* Peterborough: JNCC.

Ikeda, H., Kubota, K., Kagaya, T. and Abe, T. (2007) 'Flight capabilities and feeding habits of silphine beetles: are flightless species really “carrion beetles”?', *Ecological Research*, 22(2), pp. 237-241. doi: 10.1007/s11284-006-0012-1

Isaac, N.J.B., van Strien, A.J., August, T.A., de Zeeuw, M.P. and Roy, D.B. (2014) 'Statistics for citizen science: extracting signals of change from noisy ecological data', *Methods in Ecology and Evolution*, 5(10), pp. 1052-1060. doi: 10.1111/2041-210X.12254

Isermann, M., Koehler, H. and Mühl, M. (2010) 'Interactive effects of rabbit grazing and environmental factors on plant species-richness on dunes of Norderney', *Journal of Coastal Conservation*, 14(2), pp. 103-114. doi: 10.1007/s11852-009-0056-9

Issawi, C. (1978) 'The 1973 oil crisis and after', *Journal of Post Keynesian Economics*, 1(2), pp. 3-26. doi: 10.1080/01603477.1978.11489099

Jaffal, I., Ouldboukhitine, S.-E. and Belarbi, R. (2012) 'A comprehensive study of the impact of green roofs on building energy performance', *Renewable Energy*, 43, pp. 157-164. doi: 10.1016/j.renene.2011.12.004

Jeanne, R.L. and Morgan, R.C. (1992) 'The influence of temperature on nest site choice and reproductive strategy in a temperate zone *Polistes* wasp', *Ecological Entomology*, 17(2), pp. 135-141. doi: 10.1111/j.1365-2311.1992.tb01170.x

Jenkins, D.A. and Matthews, R.W. (2004) 'Cavity-nesting Hymenoptera in disturbed habitats of Georgia and South Carolina: nest architecture and seasonal occurrence', *Journal of the Kansas Entomological Society*, 77(3), pp. 203-214. doi: 10.2317/0212.18a.1

Jermyn, S.T. (1974) *Flora of Essex.* Colchester: Essex Naturalists' Trust.

Jessop, L. (1986) *Dung beetles and chafers: Coleoptera: Scarabaeoidea.* Rev. edn. London: Royal Entomological Society of London. Handbook for the Identification of British Insects Vol. 5, Part 11.

Jiménez-Valverde, A. and Lobo, J.M. (2007) 'Determinants of local spider (Araneidae and Thomisidae) species richness on a regional scale: climate and altitude vs. habitat structure', *Ecological Entomology*, 32(1), pp. 113-122. doi: 10.1111/j.1365-2311.2006.00848.x

Johnson, C. (1966) *Coleoptera: Clambidae.* London: Royal Entomological Society of London. Handbooks for the Identification of British Insects Vol. 4, Part 6a.

Johnson, R.A. (2000) 'Water loss in desert ants: caste variation and the effect of cuticle abrasion', *Physiological Entomology*, 25(1), pp. 48-53. doi: 10.1046/j.1365-3032.2000.00170.x

Joint Nature Conservation Committee and Department for Environment, Food & Rural Affairs (2012) *UK post-2010 Biodiversity Framework (on behalf of the Four Countries' Biodiversity Group)*. Available at: http://jncc.defra.gov.uk/pdf/UK_Post2010_Bio-Fwork.pdf (Accessed: 9 October 2017).

Jones, M.L.M., Norman, K. and Rhind, P.M. (2010) 'Topsoil inversion as a restoration measure in sand dunes, early results from a UK field-trial', *Journal of Coastal Conservation*, 14(2), pp. 139-151. doi: 10.1007/s11852-009-0072-9

Joy, N.H. (1932) *A practical handbook of British beetles.* Reprint, Faringdon: E.W. Classey Ltd, 1976.

Jungerius, P.D., Koehler, H., Kooijman, A.M., Mücher, H. and Graefe, U. (1995) 'Response of vegetation and soil ecosystem to mowing and sod removal in the coastal dunes 'Zwanenwater', the Netherlands', *Journal of Coastal Conservation*, 1(1), pp. 3-16. doi: 10.1007/BF02835557

Kadas, G. (2006) 'Rare invertebrates colonizing green roofs in London', Urban Habitats, 4(1), pp. 66-86. Available at: http://www.urbanhabitats.org/v04n01/invertebrates_full.html (Accessed: 8 October 2017).

Kamp, J., Pelster, A., Gaedicke, L., Karthäuser, J., Dieker, P. and Mantel, K. (2015) 'High nest survival and productivity of Northern Lapwings *Vanellus vanellus* breeding on urban brownfield sites', *Journal of Ornithology*, 156(1), pp. 179-190. doi: 10.1007/s10336-014-1114-0

Kaspari, M. and Joern, A. (1993) 'Prey choice by three insectivorous grassland birds: reevaluating opportunism', *Oikos*, 68(3), pp. 414-430. doi: 10.2307/3544909

Katra, I., Blumberg, D., Lavee, H. and Sarah, P. (2007) 'Topsoil moisture patterns on arid hillsides – micro-scale mapping by thermal infrared images', *Journal of Hydrology*, 334(3), pp. 359-367. doi: 10.1016/j.jhydrol.2006.10.023

Kattwinkel, M., Biedermann, R. and Kleyer, M. (2011) 'Temporary conservation for urban biodiversity', *Biological Conservation*, 144(9), pp. 2335-2343. doi: 10.1016/j.biocon.2011.06.012

Keith, D.A., Holman, L., Rodoreda, S., Lemmon, J. and Bedward, M. (2007) 'Plant functional types can predict decade-scale changes in fire-prone vegetation', *Journal of Ecology*, 95(6), pp. 1324-1337. doi: 10.1111/j.1365-2745.2007.01302.x

Kells, A.R. and Goulson, D. (2003) 'Preferred nesting sites of bumblebee queens (Hymenoptera: Apidae) in agroecosystems in the UK', *Biological Conservation*, 109(2), pp. 165-174. doi: 10.1016/S0006-3207(02)00131-3

Kessock-Philip, R., Hurst, H., Rossington, S.R. and Holloway, G.J. (2004) *Nesting ecology of Formica cunicularia*. Available at: http://publications.naturalengland.org.uk/publication/117004 (Accessed: 9 October 2017).

Ketner-Oostra, R. and Sýkora, K.V. (2004) 'Decline of lichen-diversity in calcium-poor coastal dune vegetation since the 1970s, related to grass and moss encroachment', *Phytocoenologia*, 34(4), pp. 521-549. doi: 10.1127/0340-269X/2004/0034-0521

Key, R. (2000) 'Bare ground and the conservation of invertebrates', *British Wildlife,* 11(3), pp. 183-191.

Keymer, J.E., Marquet, P.A., Velasco-Hernández, J.X. and Levin, S.A. (2000) 'Extinction thresholds and metapopulation persistence in dynamic landscapes', *The American Naturalist*, 156(5), pp. 478-494. doi: 10.1086/303407

Kirby, P. (1992) *Habitat management for invertebrates: a practical handbook.* Sandy: RSPB.

Kirk-Spriggs, A.H. (1996) *Pollen beetles: Coleoptera: Kateretidae and Nitidulidae: Meligethinae.* London: Royal Entomological Society of London. Handbooks for the Identification of British Insects Vol. 5, Part 6a.

Klausnitzer, B. (2009) *Insecta: Coleoptera: Scirtidae.* Heidelberg: Spektrum Akademischer Verlag.

Klemm, M. (1996) 'Man-made bee habitats in the anthropogenous landscape of central Europe - substitutes for threatened or destroyed riverine habitats?', in Matheson, A., Buchmann, S.L., O'Toole, C., Westrich, P. & Williams, I.H. (eds.) *The conservation of bees*. London: Academic Press, pp. 17-34.

Knisley, C.B. and Hill, J.M. (1992) 'Effects of habitat change from ecological succession and human impact on tiger beetles', *Virginia Journal of Science,* 43(1B), pp. 133-142.

Kollmann, J. (1995) 'Regeneration window for fleshy-fruited plants during scrub development on abandoned grassland', *Ecoscience*, 2(3), pp. 213-222. doi: 10.1080/11956860.1995.11682286

Komposch, C. (1996) 'Arachnological investigations on primary succession of an artificial island in southern Austria (Arachnida: Opiliones, Araneae)' (from the Proceedings of the XIIIth Congress of Arachnology, Geneva, 3-8 September 1995), *Revue Suisse de Zoologie*, 1, pp. 327-334. Available at: http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.702.8608&rep=rep1&type=pdf (Accessed: 11 October 2017).

Kooijman, A.M. and de Haan, M.W.A. (1995) 'Grazing as a measure against grass encroachment in Dutch dry dune grassland: effects on vegetation and soil', *Journal of Coastal Conservation*, 1(2), pp. 127-134. doi: 10.1007/BF02905121

Kooijman, A.M. and van der Meulen, F. (1996) 'Grazing as a control against 'grass-encroachment' in dry dune grasslands in the Netherlands', *Landscape and Urban planning*, 34(3), pp. 323-333. doi: 10.1016/0169-2046(95)00225-1

Koponen, S. (2000) 'Spiders (Araneae) on sandy islands in the southwestern archipelago of Finland', *Ekológia (Bratislava),* 19, pp. 79-85.

Kotze, D.J. and O'Hara, R.B. (2003) 'Species decline - but why? Explanations of carabid beetle (Coleoptera, Carabidae) declines in Europe', *Oecologia*, 135(1), pp. 138-148. doi: 10.1007/s00442-002-1174-3

Kraft, N.J.B., Cornwell, W.K., Webb, C.O. and Ackerly, D.D. (2007) 'Trait evolution, community assembly, and the phylogenetic structure of ecological communities', *The American Naturalist*, 170(2), pp. 271-283. doi: 10.2307/4541080

Kratochwil, A. and Klatt, M. (1989) 'Apoide Hymenopteren an ruderalstellen der stadt Freiburg i. Br. (BRD) - submediterrane faunenelemente an standorten kleinräumig hoher persistenz', *Zoologische Jahrbuch für Systematik,* 116, pp. 379-389.

Kremen, C., Williams, N.M., Bugg, R.L., Fay, J.P. and Thorp, R.W. (2004) 'The area requirements of an ecosystem service: crop pollination by native bee communities in California', *Ecology Letters*, 7(11), pp. 1109-1119. doi: 10.1111/j.1461-0248.2004.00662.x

Kronestedt, T. (1990) 'Separation of two species standing as *Alopecosa aculeata* (Clerck) by morphological, behavioural and ecological characters, with

remarks on related species in the *pulverulenta* group (Araneae, Lycosidae)', *Zoologica Scripta*, 19(2), pp. 203-225. doi: 10.1111/j.1463-6409.1990.tb00256.x

Kruess, A. and Tscharntke, T. (1994) 'Habitat fragmentation, species loss, and biological control', *Science*, 264(5165), pp. 1581-1584. doi: 10.1126/science.264.5165.1581

Kruskal, W.H. and Wallis, W.A. (1952) 'Use of ranks in one-criterion variance analysis', *Journal of the American Statistical Association*, 47(260), pp. 583-621. doi: 10.2307/2280779

Ksiazek, K., Fant, J. and Skogen, K. (2012) 'An assessment of pollen limitation on Chicago green roofs', *Landscape and Urban Planning*, 107(4), pp. 401-408. doi: 10.1016/j.landurbplan.2012.07.008

Lagerlöf, J. and Wallin, H. (1993) 'The abundance of arthropods along two field margins with different types of vegetation composition: an experimental study', *Agriculture, Ecosystems & Environment*, 43(2), pp. 141-154. doi: 10.1016/0167-8809(93)90116-7

Lai, X., Shankman, D., Huber, C., Yesou, H., Huang, Q. and Jiang, J. (2014) 'Sand mining and increasing Poyang Lake's discharge ability: a reassessment of causes for lake decline in China', *Journal of Hydrology*, 519(B), pp. 1698-1706. doi: 10.1016/j.jhydrol.2014.09.058

Laliberté, E. and Legendre, P. (2010) 'A distance-based framework for measuring functional diversity from multiple traits', *Ecology*, 91(1), pp. 299-305. doi: 10.1890/08-2244.1

Lambeets, K., Vandegehuchte, M.L., Maelfait, J.-P. and Bonte, D. (2009) 'Integrating environmental conditions and functional life-history traits for riparian arthropod conservation planning', *Biological Conservation*, 142(3), pp. 625-637. doi: 10.1016/j.biocon.2008.11.015

Lambeets, K., Vandegehuchte, M.L., Maelfait, J.P. and Bonte, D. (2008) 'Understanding the impact of flooding on trait-displacements and shifts in assemblage structure of predatory arthropods on river banks', *Journal of Animal Ecology*, 77(6), pp. 1162-1174. doi: 10.1111/j.1365-2656.2008.01443.x

Lammerts, E.J. and Grootjans, A.P. (1997) 'Nutrient deficiency in dune slack pioneer vegetation: a review', *Journal of Coastal Conservation*, 3(1), pp. 87-94. doi: 10.1007/BF02908183

Lane, S.A. and Mann, D.J. (2016) *A review of the status of the beetles of Great Britain: the stag beetles, dor beetles, dung beetles, chafers and their allies - Lucanidae, Geotrupidae, Trogidae and Scarabaeidae*. Available at: http://publications.naturalengland.org.uk/file/6560814490714112 (Accessed: 13 October 2017).

Lang, A., Filser, J. and Henschel, J.R. (1999) 'Predation by ground beetles and wolf spiders on herbivorous insects in a maize crop', *Agriculture, Ecosystems & Environment*, 72(2), pp. 189-199. doi: 10.1016/S0167-8809(98)00186-8

Langhorst, J.W. 'Rising from ruins: postindustrial sites between abandonment and engagement', *Proceedings of Open Space: People Space: an international conference on inclusive environments*, Edinburgh, Scotland, 27-29 October 2004. Available at: https://www.researchgate.net/profile/Joern_Langhorst/publication/237749013_Rising_from_Ruins_Postindustrial_Sites_Between_Abandonment_and_Engagement/links/54e2c9d80cf2c3e7d2d4822c/Rising-from-Ruins-Postindustrial-Sites-Between-Abandonment-and-Engagement.pdf (Accessed: 9 October 2017).

Langlands, P.R., Brennan, K.E.C., Framenau, V.W. and Main, B.Y. (2011) 'Predicting the post-fire responses of animal assemblages: testing a trait-based approach using spiders', *Journal of Animal Ecology*, 80(3), pp. 558-568. doi: 10.1111/j.1365-2656.2010.01795.x

Larsson, M. and Franzén, M. (2007) 'Critical resource levels of pollen for the declining bee *Andrena hattorfiana* (Hymenoptera, Andrenidae)', *Biological Conservation*, 134(3), pp. 405-414. doi: 10.1016/j.biocon.2006.08.030

Lastrucci, L., Landi, M. and Angiolini, C. (2010) 'Vegetation analysis on wetlands in a Tuscan agricultural landscape (central Italy)', *Biologia*, 65(1), pp. 54-68. doi: 10.2478/s11756-009-0213-5

Lavorel, S. and Garnier, É. (2002) 'Predicting changes in community composition and ecosystem functioning from plant traits: revisiting the Holy Grail', *Functional Ecology*, 16(5), pp. 545-556. doi: 10.1046/j.1365-2435.2002.00664.x

Lawton, J.H. (1983) 'Plant architecture and the diversity of phytophagous insects', *Annual Review of Entomology*, 28(1), pp. 23-39. doi: 10.1146/annurev.en.28.010183.000323

Lawton, J.H. (1999) 'Are there general laws in ecology?', *Oikos*, 84(2), pp. 177-192. doi: 10.2307/3546712

Lawton, J.H., Brotherton, P.N.M., Brown, V.K., Elphick, C., Fitter, A.H., Forshaw, J., Haddow, R.W., Hilborne, S., Leafe, R.N. and Mace, G.M. (2010) *Making space for nature: a review of England's wildlife sites and ecological network*. Available at: http://webarchive.nationalarchives.gov.uk/20130402170324/http://archive.defra.gov.uk/environment/biodiversity/documents/201009space-for-nature.pdf (Accessed: 9 October 2017).

Lee, J.A. and Greenwood, B. (1976) 'The colonisation by plants of calcareous wastes from the salt and alkali industry in Cheshire, England', *Biological Conservation*, 10(2), pp. 131-149. doi: 10.1016/0006-3207(76)90057-4

Legendre, P. and Gallagher, E.D. (2001) 'Ecologically meaningful transformations for ordination of species data', *Oecologia*, 129(2), pp. 271-280. doi: 10.1007/s004420100716

Legendre, P., Galzin, R. and Harmelin-Vivien, M.L. (1997) 'Relating behaviour to habitat: solutions to the fourth-corner problem', *Ecology*, 78(2), pp. 547-562. doi: 10.1890/0012-9658(1997)078[0547:RBTHST]2.0.CO;2

Lenda, M., Skórka, P., Moroń, D., Rosin, Z.M. and Tryjanowski, P. (2012) 'The importance of the gravel excavation industry for the conservation of grassland butterflies', *Biological Conservation*, 148(1), pp. 180-190. doi: 10.1016/j.biocon.2012.01.014

Levi, D. and Kocher, S. (2006) 'The use of coastal brownfields as nature preserves', *Environment and Behavior*, 38(6), pp. 802-819. doi: 10.1177/0013916505285217

Liebergesell, M., Reu, B., Stahl, U., Freiberg, M., Welk, E., Kattge, J., Cornelissen, J.H.C., Peñuelas, J. and Wirth, C. (2016) 'Functional resilience against climate-driven extinctions – comparing the functional diversity of European and North American tree floras', *PLOS ONE,* 11(2), p. e0148607.

Lin, Y.C., James, R. and Dolman, P.M. (2005) 'Are pitfalls biased? A comparison of carabid composition from pitfall trapping and hand searching in forest habitats', *British Journal of Entomology and Natural History,* 18(1), pp. 17-25.

Lindsay, E.A. and Cunningham, S.A. (2009) 'Livestock grazing exclusion and microhabitat variation affect invertebrates and litter decomposition rates in woodland remnants', *Forest Ecology and Management*, 258(2), pp. 178-187. doi: 10.1016/j.foreco.2009.04.005

Littlewood, N.A., Dennis, P., Pakeman, R.J. and Woodin, S.J. (2006) 'Moorland restoration aids the reassembly of associated phytophagous insects', *Biological Conservation*, 132(3), pp. 395-404. doi: 10.1016/j.biocon.2006.04.033

Littlewood, N.A., Pakeman, R.J. and Woodin, S.J. (2009) 'Isolation of habitat patches limits colonisation by moorland Hemiptera', *Journal of Insect Conservation*, 13(1), pp. 29-36. doi: 10.1007/s10841-007-9114-5

Littlewood, N.A., Stewart, A.J.A. and Woodcock, B.A. (2012) 'Science into practice – how can fundamental science contribute to better management of grasslands for invertebrates?', *Insect Conservation and Diversity*, 5(1), pp. 1-8. doi: 10.1111/j.1752-4598.2011.00174.x

Lohse, G.A. and Klausnitzer, B. (1992) *Die Käfer Mitteleuropas: 2. Supplementband mit katalogteil.* Krefeld, Germany: Goecke & Evers Verlag.

London Development Agency (England) (2007) *London brownfield sites review: Stage 1*. Available at: https://www.london.gov.uk/moderngov/Data/Environment%20Committee/20070308/Agenda/3%20Appendix%201%20PDF.pdf (Accessed: 10 October 2017).

London Wildlife Trust (2002) *Brownfield? Greenfield? A threat to London's unofficial countryside*. Available at: http://www.wildlondon.org.uk/sites/default/files/files/Brownfield%20Greenfield%20-%20the%20threat%20to%20Londons%20unofficial%20countryside.pdf (Accessed: 9 October 2017).

Longo, A. and Campbell, D. (2017) 'The determinants of brownfields redevelopment in England', *Environmental and Resource Economics*, 67(2), pp. 261-283. doi: 10.1007/s10640-015-9985-y

Lönnberg, L. and Jonsell, M. (2012) 'Sand pits as habitats for beetles (Coleoptera): does area affect species number and composition?', *Biodiversity and Conservation*, 21(3), pp. 853-874. doi: 10.1007/s10531-012-0225-2

Lott, D. (2001) 'Ground beetles and rove beetles associated with temporary ponds in England' (from the European temporary ponds conference, Birmingham, February 2001), *Freshwater Forum*, 17(1), pp. 40-53. Available at: https://core.ac.uk/download/pdf/11020167.pdf (Accessed: 11 October 2017).

Lott, D.A. (1998) *The semi-aquatic-habitats of terrestrial Coleoptera in a lowland river floodplain.* PhD thesis. University of Newcastle upon Tyne. Available at: http://ethos.bl.uk/OrderDetails.do?uin=uk.bl.ethos.287803 (Accessed: 8 October 2017).

Lott, D.A. (2009) *The Staphylinidae (rove beetles) of Britain and Ireland: Part 5: Scaphidiinae, Piestinae, Oxytelinae.* St. Albans: Royal Entomological Society. Handbooks for the Identification of British Insects Vol. 12, Part 5.

Lott, D.A. and Anderson, R. (2011) *The Staphylinidae (rove beetles) of Britain and Ireland: Parts 7 and 8: Oxyporinae, Steninae, Euaesthetinae, Pseudopsinae, Paederinae, Staphylininae.* St Albans: Royal Entomological Society. Handbooks for the Identification of British Insects, Vol. 12, Part 7.

Lucht, W. and Klausnitzer, B. (1998) *Die Käfer Mitteleuropas: 4. Supplementband.* Krefeld, Germany: Goecke & Evers Verlag.

Lückmann, J. and Assmann, T. (2006) 'Reproductive biology and strategies of nine meloid beetles from Central Europe (Coleoptera: Meloidae)', *Journal of Natural History*, 39(48), pp. 4101-4125. doi: 10.1080/00222930500533823

Luff, M.L. (1965) 'The morphology and microclimate of *Dactylis glomerata* tussocks', *Journal of Ecology,* 53(3), pp. 771-787.

Luff, M.L. (1966) 'The abundance and diversity of the beetle fauna of grass tussocks', *Journal of Animal Ecology*, 35(1), pp. 189-208. doi: 10.2307/2696

Luff, M.L. (1975) 'Some features influencing the efficiency of pitfall traps', *Oecologia*, 19, pp. 345-357. doi: 10.1007/BF00348110

Luff, M.L. (2007) *The Carabidae (ground beetles) of Britain and Ireland.* 2nd edn. St. Albans: Royal Entomological Society. Hanbooks for the Identification of British Insects Vol. 4, Part 2.

Lush, M.J., Shepherd, P., Harvey, P., Lush, C.E. and Griffiths, R. (2014) *Definition and mapping of open mosaic habitats on previously developed land: phase 2 testing methods and developing the habitat inventory (WC0795)*. Talgarth, Powys: ExeGesIS Spatial Data Management.

Lyons, G. and Pryor, S. (2014) *Monitoring of the invertebrates of the turf-stripped areas of Iping Common with an emphasis on Heath Tiger Beetles - a 2013 baseline*. Henfield: Sussex Wildlife Trust.

Macadam, C., Bairner, S. and Cathrine, C. (2013) *Open Mosaic Habitats on previously developed land: survey and recommendations for habitat planning and management in Scotland*. Available at: www.snh.org.uk/pdfs/publications/commissioned_reports/606.pdf (Accessed: 9 October 2017).

MacArthur, R.H. and Wilson, E.O. (1967) *The theory of island biogeography.* Princeton, NJ: Princeton University Press.

MacIvor, J.S. (2016) 'Building height matters: nesting activity of bees and wasps on vegetated roofs', *Israel Journal of Ecology & Evolution*, 62(1-2), pp. 88-96. doi: 10.1080/15659801.2015.1052635

MacIvor, J.S. (2017) 'Cavity-nest boxes for solitary bees: a century of design and research', *Apidologie*, 48(3), pp. 311-327. doi: 10.1007/s13592-016-0477-z

MacIvor, J.S. and Lundholm, J. (2011) 'Performance evaluation of native plants suited to extensive green roof conditions in a maritime climate', *Ecological Engineering*, 37(3), pp. 407-417. doi: 10.1016/j.ecoleng.2010.10.004

MacIvor, J.S. and Packer, L. (2015) ''Bee hotels' as tools for native pollinator conservation: a premature verdict?', *PLOS ONE*, 10(3), e0122126 (13pp). doi: 10.1371/journal.pone.0122126

Maddock, A. (ed.) (2011) *UK Biodiversity Action Plan: Priority Habitat descriptions.* Rev. edn. Available at: http://jncc.defra.gov.uk/PDF/UKBAP_PriorityHabitatDesc-Rev2011.pdf (Accessed: 11 October 2017).

Madre, F., Vergnes, A., Machon, N. and Clergeau, P. (2013) 'A comparison of 3 types of green roof as habitats for arthropods', *Ecological Engineering*, 57, pp. 109-117. doi: 10.1016/j.ecoleng.2013.04.029

(2014) 'Green roofs as habitats for wild plant species in urban landscapes: first insights from a large-scale sampling', *Landscape and Urban Planning*, 122, pp. 100-107. doi: 10.1016/j.landurbplan.2013.11.012

Magura, T., Tóthmérész, B. and Elek, Z. (2005) 'Impacts of leaf-litter addition on carabids in a conifer plantation', *Biodiversity and Conservation*, 14(2), pp. 475-491. doi: 10.1007/s10531-004-7307-8

Magura, T., Tóthmérész, B. and Molnár, T. (2001) 'Forest edge and diversity: carabids along forest-grassland transects', *Biodiversity and Conservation*, 10(2), pp. 287-300. doi: 10.1023/A:1008967230493

Magurran, A.E. (2004) *Measuring biological diversity.* Oxford: Blackwell Publishing.

Majerus, M., Kearns, P. and Allington, S. (1989) *Ladybirds.* Slough: The Richmond Publishing Co. Ltd. Naturalists' Handbooks 10.

Manning, A.D., Fischer, J. and Lindenmayer, D.B. (2006) 'Scattered trees are keystone structures – implications for conservation', *Biological Conservation*, 132(3), pp. 311-321. doi: 10.1016/j.biocon.2006.04.023

Mantilla-Contreras, J., Schirmel, J. and Zerbe, S. (2011) 'Influence of soil and microclimate on species composition and grass encroachment in heath succession', *Journal of Plant Ecology*, 5(3), pp. 249-259. doi: 10.1093/jpe/rtr031

Martinsen, G.D., Cushman, J.H. and Whitham, T.G. (1990) 'Impact of pocket gopher disturbance on plant species diversity in a shortgrass prairie community', *Oecologia*, 83(1), pp. 132-138. doi: 10.1007/BF00324644

Mason, N.W.H., Mouillot, D., Lee, W.G. and Wilson, J.B. (2005) 'Functional richness, functional evenness and functional divergence: the primary components of functional diversity', *Oikos*, 111(1), pp. 112-118. doi: 10.1111/j.0030-1299.2005.13886.x

Mathey, J., Arndt, T., Banse, J. and Rink, D. (2016) 'Public perception of spontaneous vegetation on brownfields in urban areas - results from surveys in Dresden and Leipzig (Germany)'. To be published in *Urban Forestry & Urban Greening* [Preprint]. doi: 10.1016/j.ufug.2016.10.007.

Maun, M.A. (1994) 'Adaptations enhancing survival and establishment of seedlings on coastal dune systems', *Plant Ecology*, 111(1), pp. 59-70. doi: 10.1007/BF00045577

McCorquodale, D.B. (1989) 'Soil softness, nest initiation and nest sharing in the wasp, *Cerceris antipodes* (Hymenoptera: Sphecidae)', *Ecological Entomology*, 14(2), pp. 191-196. doi: 10.1111/j.1365-2311.1989.tb00769.x

McCracken, D.I. and Tallowin, J.R. (2004) 'Swards and structure: the interactions between farming practices and bird food resources in lowland grasslands', *Ibis*, 146(S2), pp. 108-114. doi: 10.1111/j.1474-919X.2004.00360.x

McGeoch, M.A. (1998) 'The selection, testing and application of terrestrial insects as bioindicators', *Biological Reviews*, 73(2), pp. 181-201. doi: 10.1111/j.1469-185X.1997.tb00029.x

McGill, B.J., Enquist, B.J., Weiher, E. and Westoby, M. (2006) 'Rebuilding community ecology from functional traits', *Trends in Ecology & Evolution*, 21(4), pp. 178-185. doi: 10.1016/j.tree.2006.02.002

McIntyre, S., Lavorel, S., Landsberg, J. and Forbes, T.D.A. (1999) 'Disturbance response in vegetation – towards a global perspective on functional traits', *Journal of Vegetation Science*, 10(5), pp. 621-630. doi: 10.2307/3237077

McKenna, D.D., Farrell, B.D., Caterino, M.S., Farnum, C.W., Hawks, D.C., Maddison, D.R., Seago, A.E., Short, A.E.Z., Newton, A.F. and Thayer, M.K. (2015) 'Phylogeny and evolution of Staphyliniformia and Scarabaeiformia: forest litter as a stepping stone for diversification of nonphytophagous beetles', *Systematic Entomology*, 40(1), pp. 35-60. doi: 10.1111/syen.12093

McKinney, M.L. (2008) 'Effects of urbanization on species richness: a review of plants and animals', *Urban Ecosystems*, 11(2), pp. 161-176. doi: 10.1007/s11252-007-0045-4

Meijer, J. (1977) 'The immigration of spiders (Araneida) into a new polder', *Ecological Entomology*, 2(1), pp. 81-90. doi: 10.1111/j.1365-2311.1977.tb00867.x

Melbourne, B.A. (1999) 'Bias in the effect of habitat structure on pitfall traps: an experimental evaluation', *Austral Ecology*, 24(3), pp. 228-239. doi: 10.1046/j.1442-9993.1999.00967.x

Melnychuk, N.A., Olfert, O., Youngs, B. and Gillott, C. (2003) 'Abundance and diversity of Carabidae (Coleoptera) in different farming systems', *Agriculture, Ecosystems & Environment*, 95(1), pp. 69-72. doi: 10.1016/S0167-8809(02)00119-6

Merivee, E., Ploomi, A., Rahi, M., Bresciani, J., Ravn, H.P., Luik, A. and Sammelselg, V. (2002) 'Antennal sensilla of the ground beetle *Bembidion properans* Steph. (Coleoptera, Carabidae)', *Micron,* 33(5), pp. 429-440.

Merrett, P. (1990) *A review of the Nationally Notable spiders of Great Britain. CSD Contract Surveys No. 127.* Peterborough: Nature Conservancy Council.

Met Office Press Office (2016) *Early June 2016 statistics*. Available at: https://blog.metoffice.gov.uk/2016/06/29/early-june-2016-statistics/ (Accessed: 18 May 2017).

Millett, J. and Edmondson, S. (2013) 'The impact of 36 years of grazing management on vegetation dynamics in dune slacks', *Journal of Applied Ecology*, 50(6), pp. 1367-1376. doi: 10.1111/1365-2664.12113

Molineux, C.J., Gange, A.C., Connop, S.P. and Newport, D.J. (2015) 'Using recycled aggregates in green roof substrates for plant diversity', *Ecological Engineering*, 82(Supplement C), pp. 596-604. doi: https://doi.org/10.1016/j.ecoleng.2015.05.036

Moretti, M., Duelli, P. and Obrist, M.K. (2006) 'Biodiversity and resilience of arthropod communities after fire disturbance in temperate forests', *Oecologia*, 149(2), pp. 312-327. doi: 10.1007/s00442-006-0450-z

Moroń, D., Grześ, I.M., Skorka, P., Szentgyörgyi, H., Laskowski, R., Potts, S.G. and Woyciechowski, M. (2012) 'Abundance and diversity of wild bees along gradients of heavy metal pollution', *Journal of Applied Ecology*, 49(1), pp. 118-125. doi: 10.1111/j.1365-2664.2011.02079.x

Morris, M.G. (1990) 'The Hemiptera of two sown calcareous grasslands. II. Differences between treatments', *Journal of Applied Ecology*, 27(2), pp. 379-393. doi: 10.2307/2404288

Morris, M.G. (2000) 'The effects of structure and its dynamics on the ecology and conservation of arthropods in British grasslands', *Biological Conservation*, 95(2), pp. 129-142. doi: 10.1016/S0006-3207(00)00028-8

Morris, M.G. (2002) *True weevils (Part I): Coleoptera: Curculionidae (subfamilies Raymondionyminae to Smicronychinae).* St. Albans: Royal Entomological Society. Handbooks for the Identification of British Insects Vol. 5, Part 17b.

Morris, M.G., Thomas, J.A., Ward, L.K., Snazell, R.G., Pywell, R.F., Stevenson, M.J. and Webb, N.R. (1994) 'Re-creation of early-successional stages for threatened butterflies - an ecological engineering approach', *Journal of Environmental Management*, 42(2), pp. 119-135. doi: 10.1006/jema.1994.1065

Morris, R.K.A., Alonso, I., Jefferson, R.G. and Kirby, K.J. (2006) 'The creation of compensatory habitat - can it secure sustainable development?', *Journal for Nature Conservation*, 14(2), pp. 106-116. doi: 10.1016/j.jnc.2006.01.003

Morrocco, S.M. and Ballantyne, C.K. (2008) 'Footpath morphology and terrain sensitivity on high plateaux: the Mamore Mountains, Western Highlands of Scotland', *Earth Surface Processes and Landforms*, 33(1), pp. 40-54. doi: 10.1002/esp.1525

Morse, D.H. (1993) 'Some determinants of dispersal by crab spiderlings', *Ecology*, 74(2), pp. 427-432. doi: 10.2307/1939304

Mortimer, S.R., Booth, R.G., Harris, S.J. and Brown, V.K. (2002) 'Effects of initial site management on the Coleoptera assemblages colonising newly established chalk grassland on ex-arable land', *Biological Conservation*, 104(3), pp. 301-313. doi: 10.1016/S0006-3207(01)00195-1

Mortimer, S.R., Hollier, J.A. and Brown, V.K. (1998) 'Interactions between plant and insect diversity in the restoration of lowland calcareous grasslands in southern Britain', *Applied Vegetation Science*, 1(1), pp. 101-114. doi: 10.2307/1479089

Mortimer, S.R., Turner, A.J., Brown, V.K., Fuller, R.J., Good, J.E.G., Bell, S.A., Stevens, P.A., Norris, D., Bayfield, N.G. and Ward, L.K. (2000) *The nature conservation value of scrub in Britain*. Available at: http://jncc.defra.gov.uk/page-2445 (Accessed: 9 October 2017).

Morton, D., Rowland, C., Wood, C., Meek, L., Marston, C., Smith, G., Wadsworth, R. and Simpson, I. (2011) *Countryside Survey: final report for LCM2007 - the new UK land cover map*. Available at: https://www.ceh.ac.uk/sites/default/files/LCM2007%20Final%20Report.pdf (Accessed: 9 October 2017).

Moser, K., Ahn, C. and Noe, G. (2007) 'Characterization of microtopography and its influence on vegetation patterns in created wetlands', *Wetlands*, 27(4), pp. 1081-1097. doi: 10.1672/0277-5212(2007)27[1081:COMAII]2.0.CO;2

Mrzljak, J. and Wiegleb, G. (2000) 'Spider colonization of former brown coal mining areas - time or structure dependent?', *Landscape and Urban Planning*, 51(2), pp. 131-146. doi: 10.1016/S0169-2046(00)00104-3

Müller, J. (2004) 'Cost-benefit ratio and empirical examination of the acceptance of heathland maintenance in the Lueneburg Heath Nature Reserve', *Journal of Environmental Planning and Management*, 47(5), pp. 757-771. doi: 10.1080/0964056042000274461

Nash, C., Clough, J., Gedge, D., Lindsay, R., Newport, D., Ciupala, M.A. and Connop, S. (2016) 'Initial insights on the biodiversity potential of biosolar roofs: a London Olympic Park green roof case study', *Israel Journal of Ecology & Evolution*, 62(1-2), pp. 74-87. doi: 10.1080/15659801.2015.1045791

Nassauer, J.I. and Raskin, J. (2014) 'Urban vacancy and land use legacies: a frontier for urban ecological research, design, and planning', *Landscape and Urban Planning*, 125, pp. 245-253. doi: 10.1016/j.landurbplan.2013.10.008

Natural England (2008) *State of the Natural Environment*. Available at: http://publications.naturalengland.org.uk/publication/31043 (Accessed: 9 October 2017).

Natural Environment and Rural Communities Act 2006, c. 16. Available at: https://www.legislation.gov.uk/ukpga/2006/16/contents (Accessed: 8 October 2017).

Niemelä, J. and Spence, J.R. (1999) 'Dynamics of local expansion by an introduced species: *Pterostichus melanarius* Ill. (Coleoptera, Carabidae) in Alberta, Canada', *Diversity and Distributions*, 5(4), pp. 121-127. doi: 10.1046/j.1472-4642.1999.00048.x

Niemeyer, M., Niemeyer, T., Fottner, S., Härdtle, W. and Mohamed, A. (2007) 'Impact of sod-cutting and choppering on nutrient budgets of dry heathlands', *Biological Conservation*, 134(3), pp. 344-353. doi: 10.1016/j.biocon.2006.07.013

Nieto, A., Roberts, S.P., Kemp, J., Rasmont, P., Kuhlmann, M., García Criado, M., Biesmeijer, J., Bogusch, P., Dathe, H. and De la Rúa, P. (2014) *European red list of bees*. Available at: http://ec.europa.eu/environment/nature/conservation/species/redlist/downloads/European_bees.pdf (Accessed: 18 October 2017).

Nordén, B., Ryberg, M., Götmark, F. and Olausson, B. (2004) 'Relative importance of coarse and fine woody debris for the diversity of wood-inhabiting fungi in temperate broadleaf forests', *Biological Conservation*, 117(1), pp. 1-10. doi: 10.1016/S0006-3207(03)00235-0

Novotná, L. and Šťastná, P. (2013) 'Ground beetles (Carabidae) on quarry terraces in the vicinity of Brno (Czech Republic)', *Acta Universitatis Agriculturae et Silviculturae Mendelianae Brunensis*, 60(3), pp. 147-154. doi: 10.11118/actaun201260030147

O'Callaghan, M.J., Hannah, D.M., Williams, M. and Sadler, J.P. (2013) 'Exposed riverine sediments (ERS) in England and Wales: distribution, controls and management', *Aquatic Conservation: Marine and Freshwater Ecosystems*, 23(6), pp. 924-938. doi: 10.1002/aqc.2376

Obrist, M.K. and Duelli, P. (1996) 'Trapping efficiency of funnel-and cup-traps for epigeal arthropods', *Mitteilungen der Schweizerischen Entomologischen Gesellschaft*, 69, pp. 361-369. doi: 10.5169/seals-402638

Ödman, A.M., Mårtensson, L.M., Sjöholm, C. and Olsson, P.A. (2011) 'Immediate responses in soil chemistry, vegetation and ground beetles to soil perturbation when implemented as a restoration measure in decalcified sandy grassland', *Biodiversity and Conservation*, 20(13), pp. 3039-3058. doi: 10.1007/s10531-011-0108-y

Office of the Deputy Prime Minister (England) (2005) *Planning for mixed communities: a consultation paper on a proposed change to Planning Policy Guidance Note 3: Housing*. Available at: https://publications.parliament.uk/pa/cm200405/cmhansrd/vo050124/wmstext/50124m01.htm (Accessed: 9 October 2017).

Olff, H., Huisman, J. and Van Tooren, B.F. (1993) 'Species dynamics and nutrient accumulation during early primary succession in coastal sand dunes', *Journal of Ecology*, 81(4), pp. 693-706. doi: 10.2307/2261667

Oliver, L., Ferber, U., Grimski, D., Millar, K. and Nathanail, P. 'The scale and nature of European brownfields', *Proceedings of CABERNET 2005: the international conference on managing urban land*, Belfast, Northern Ireland, 13-15 April 2005, pp. 274-281. Available at: https://www.researchgate.net/profile/Uwe_Ferber/publication/228789048_The_Scale_and_Nature_of_European_Brownfield/links/5469e8300cf20dedafd20077/The-Scale-and-Nature-of-European-Brownfield.pdf (Accessed: 9 October 2017).

Oliver, T.H., Gillings, S., Pearce-Higgins, J.W., Brereton, T., Crick, H.Q.P., Duffield, S.J., Morecroft, M.D. and Roy, D.B. (2017) 'Large extents of intensive land use limit community reorganization during climate warming', *Global Change Biology*, 23(6), pp. 2272-2283. doi: 10.1111/gcb.13587

Olly, L.M., Bates, A.J., Sadler, J.P. and Mackay, R. (2011) 'An initial experimental assessment of the influence of substrate depth on floral assemblage for extensive green roofs', *Urban Forestry & Urban Greening*, 10(4), pp. 311-316. doi: 10.1016/j.ufug.2011.07.005

Olsson, P.A., Sjöholm, C. and Ödman, A.M. (2014) 'Soil disturbance favours threatened beetle species in sandy grasslands', *Journal of Insect Conservation*, 18(5), pp. 827-835. doi: 10.1007/s10841-014-9689-6

Orford, K.A., Vaughan, I.P. and Memmott, J. (2015) 'The forgotten flies: the importance of non-syrphid Diptera as pollinators', *Proceedings of the Royal Society of London B: Biological Sciences*, 282(1805), 20142934 (6pp). doi: 10.1098/rspb.2014.2934

Oxbrough, A.G., Gittings, T., O'Halloran, J., Giller, P.S. and Smith, G.F. (2005) 'Structural indicators of spider communities across the forest plantation cycle', *Forest Ecology and Management*, 212(1-3), pp. 171-183. doi: 10.1016/j.foreco.2005.03.040

Pagano, M.A. and Bowman, A.O.M. (2000) *Vacant land in cities: an urban resource*. Available at: https://www.brookings.edu/wp-content/uploads/2016/06/paganofinal.pdf (Accessed: 9 October 2017).

Pakeman, R.J. (2014) 'Functional trait metrics are sensitive to the completeness of the species' trait data?', *Methods in Ecology and Evolution*, 5(1), pp. 9-15. doi: 10.1111/2041-210X.12136

Paliy, O. and Shankar, V. (2016) 'Application of multivariate statistical techniques in microbial ecology', *Molecular Ecology,* 25(5), pp. 1032-1057.

Pankhurst, H.J. (2012) *Green Infrastructure: mainstreaming the concept: understanding and applying the principles of green infrastructure in South*

Worcestershire. MSc thesis. University of Gloucestershire. Available at: http://publications.naturalengland.org.uk/file/275811 (Accessed: 8 October 2017).

Pape, T. (1987) *The Sarcophagidae (Diptera) of Fennoscandia and Denmark.* Copenhagen: E.J. Brill.

Paradis, E., Claude, J. and Strimmer, K. (2004) APE: analyses of phylogenetics and evolution in R language (Version 4.1) [R package]. Available at: https://cran.r-project.org/web/packages/ape/index.html (Accessed: 7 October 2017).

Passingham, E. (2013) 'UK conservation of streaked bombardier *Brachinus sclopeta* (Fabricius 1792) (Coleoptera: Carabidae)', *Essex Naturalist,* 30, pp. 84-91.

Peacock, E.R. (1993) *Adults and larvae of hide, larder and carpet beetles and their relatives (Coleoptera: Dermestidae) and of derodontid beetles (Coleoptera: Derodontidae).* London: Royal Entomological Society of London. Handbooks for the Identification of British Insects Vol. 5, Part 3.

Pearce, H. and Walters, C.L. (2012) 'Do green roofs provide habitat for bats in urban areas?', *Acta Chiropterologica*, 14(2), pp. 469-478. doi: 10.3161/150811012X661774

Pedley, S.M. and Dolman, P.M. (2014) 'Multi-taxa trait and functional responses to physical disturbance', *Journal of Animal Ecology*, 83(6), pp. 1542-1552. doi: 10.1111/1365-2656.12249

Pedley, S.M., Franco, A.M.A., Pankhurst, T. and Dolman, P.M. (2013) 'Physical disturbance enhances ecological networks for heathland biota: a multiple taxa experiment', *Biological Conservation*, 160, pp. 173-182. doi: 10.1016/j.biocon.2013.01.006

Peng, R.K., Sutton, S.L. and Fletcher, C.R. (1992) 'Spatial and temporal distribution patterns of flying Diptera', *Journal of Zoology*, 228(2), pp. 329-340. doi: 10.1111/j.1469-7998.1992.tb04612.x

Penney, M.M. (1966) 'Studies on certain aspects of the ecology of *Nebria brevicollis* (F.) (Coleoptera, Carabidae)', *Journal of Animal Ecology*, 35(3), pp. 505-512. doi: 10.2307/2488

Penny Anderson Associates (2005) *Aculeate Hymenoptera and Diptera of Chafford Hundred: a comparison between 1995 and 2005*. Unpublished report.

Peterson, J.H. and Roitberg, B.D. (2006) 'Impact of resource levels on sex ratio and resource allocation in the solitary bee, *Megachile rotundata*', *Environmental Entomology*, 35(5), pp. 1404-1410. doi: 10.1093/ee/35.5.1404

Peterson, J.H., Roitberg, B.D. and Peterson, J.H. (2006) 'Impacts of flight distance on sex ratio and resource allocation to offspring in the leafcutter bee, Megachile rotundata', *Behavioral Ecology and Sociobiology*, 59(5), pp. 589-596. doi: 10.1007/s00265-005-0085-9

Pétillon, J., Georges, A., Canard, A., Lefeuvre, J.-C., Bakker, J.P. and Ysnel, F. (2008) 'Influence of abiotic factors on spider and ground beetle communities in different salt-marsh systems', *Basic and Applied Ecology*, 9(6), pp. 743-751. doi: 10.1016/j.baae.2007.08.007

Pétillon, J., Ysnel, F., Canard, A. and Lefeuvre, J.-C. (2005) 'Impact of an invasive plant (*Elymus athericus*) on the conservation value of tidal salt marshes in western France and implications for management: responses of spider populations', *Biological Conservation*, 126(1), pp. 103-117. doi: 10.1016/j.biocon.2005.05.003

Picaud, F. and Petit, D.P. (2007) 'Primary succession of Orthoptera on mine tailings: role of vegetation', *Annales de la Société Entomologique de France*, 43(1), pp. 69-79. doi: 10.1080/00379271.2007.10697496

Pik, A.J., Dangerfield, J.M., Bramble, R.A., Angus, C. and Nipperess, D.A. (2002) 'Ths use of invertebrates to assess small-scale habitat heterogeneity and its application to restoration practices', *Environmental Monitoring and Assessment*, 75(2), pp. 179-199. doi: 10.1023/A:101444403

Podani, J. (1999) 'Extending Gower's general coefficient of similarity to ordinal characters', *Taxon*, 48(2), pp. 331-340. doi: 10.2307/1224438

Polidori, C., Rubichi, A., Barbieri, V., Trombino, L. and Donegana, M. (2010) 'Floral resources and nesting requirements of the ground-nesting social bee, *Lasioglossum malachurum* (Hymenoptera: Halictidae), in a Mediterranean semiagricultural landscape', *Psyche*, 2010, 851947 (11pp). doi: 10.1155/2010/851947

Pollet, M. and Grootaert, P. (1996) 'An estimation of the natural value of dune habitats using Empidoidea (Diptera)', *Biodiversity and Conservation*, 5(7), pp. 859-880. doi: 10.1007/BF00054739

Port of London Authority (2014) *Port of London Authority: baseline document for maintenance dredging*. Available at: http://pla.co.uk/assets/r2238afinalmdpbaselinedocument7oct2014.compressed1.pdf (Accessed: 9 October 2017).

Potts, S. and Willmer, P. (1997) 'Abiotic and biotic factors influencing nest-site selection by *Halictus rubicundus*, a ground-nesting halictine bee', *Ecological Entomology*, 22(3), pp. 319-328. doi: 10.1046/j.1365-2311.1997.00071.x

Potts, S.G., Vulliamy, B., Dafni, A., Ne'eman, G. and Willmer, P. (2003) 'Linking bees and flowers: how do floral communities structure pollinator communities?', *Ecology*, 84(10), pp. 2628-2642. doi: 10.1890/02-0136

Potts, S.G., Vulliamy, B., Roberts, S., O'Toole, C., Dafni, A., Ne'eman, G. and Willmer, P. (2005) 'Role of nesting resources in organising diverse bee communities in a Mediterranean landscape', *Ecological Entomology*, 30(1), pp. 78-85. doi: 10.1111/j.0307-6946.2005.00662.x

Potts, S.G. and Willmer, P. (1998) 'Compact housing in built-up areas: spatial patterning of nests in aggregations of a ground-nesting bee', *Ecological Entomology*, 23(4), pp. 427-432. doi: 10.1046/j.1365-2311.1998.00160.x

Prach, K. and Pyšek, P. (2001) 'Using spontaneous succession for restoration of human-disturbed habitats: experience from Central Europe', *Ecological Engineering*, 17(1), pp. 55-62. doi: 10.1016/S0925-8574(00)00132-4

Prach, K., Řehounková, K., Řehounek, J. and Konvalinková, P. (2011) 'Ecological restoration of central European mining sites: a summary of a multi-site analysis', *Landscape Research*, 36(2), pp. 263-268. doi: 10.1080/01426397.2010.547571

Prather, C.M., Pelini, S.L., Laws, A., Rivest, E., Woltz, M., Bloch, C.P., Del Toro, I., Ho, C.K., Kominoski, J. and Newbold, T.A. (2013) 'Invertebrates, ecosystem services and climate change', *Biological Reviews*, 88(2), pp. 327-348. doi: 10.1111/brv.12002

Praz, C.J., Müller, A. and Dorn, S. (2008) 'Specialized bees fail to develop on non-host pollen: do plants chemically protect their pollen', *Ecology*, 89(3), pp. 795-804. doi: 10.1890/07-0751.1

Prentice, I.C. (1977) 'Non-metric ordination methods in ecology', *Journal of Ecology,* 65(1), pp. 85-94.

Prévosto, B., Kuiters, L., Bernhardt-Römermann, M., Dölle, M., Schmidt, W., Hoffmann, M., Van Uytvanck, J., Bohner, A., Kreiner, D. and Stadler, J. (2011) 'Impacts of land abandonment on vegetation: successional pathways in European habitats', *Folia Geobotanica*, 46(4), pp. 303-325. doi: 10.1007/s12224-010-9096-z

Pulliam, H.R. (1988) 'Sources, sinks, and population regulation', *The American Naturalist*, 132(5), pp. 652-661. doi: 10.1086/284880

Purtauf, T., Dauber, J. and Wolters, V. (2005) 'The response of carabids to landscape simplification differs between trophic groups', *Oecologia*, 142(3), pp. 458-464. doi: 10.1007/s00442-004-1740-y

Quinn, J.F. and Robinson, G.R. (1987) 'The Effects of Experimental Subdivision on Flowering Plant Diversity in a California Annual Grassland', *Journal of Ecology*, 75(3), pp. 837-855. doi: 10.2307/2260209

R Core Team (2017) R: a language and environment for statistical computing (Version 3.3.3) [Computer program]. Available at: https://www.R-project.org/ (Accessed: 7 October 2017). R Foundation for Statistical Computing, Vienna, Austria.

Rahman, M.L., Tarrant, S., McCollin, D. and Ollerton, J. (2015) 'Vegetation cover and grasslands in the vicinity accelerate development of carabid beetle assemblages on restored landfill sites', *Zoology and Ecology,* 25(4), pp. 347-354.

Rahman, M.L., Tarrant, S., Ollerton, J. and McCollin, D. (2016) 'Effect of soil conditions and landscape factors on macro-snail communities in newly created grasslands of restored landfill sites in the UK', *Zoology and Ecology,* 26(4), pp. 275-281.

Ravenscroft, N.O.M. and Warren, M.S. (1996) *Species Action Plan: Northern Brown Argus Aricia artaxerxes*. Available at: https://butterfly-conservation.org/files/northern-brown-argus-action-plan.doc (Accessed: 9 October 2017).

Raw, A. (1972) 'The biology of the solitary bee *Osmia rufa* (L.) (Megachilidae)', *Transactions of the Royal Entomological Society of London*, 124(3), pp. 213-229. doi: 10.1111/j.1365-2311.1972.tb00364.x

Redhead, J., Cuevas-Gonzales, M., Smith, G., Gerard, F. and Pywell, R. (2012) 'Assessing the effectiveness of scrub management at the landscape scale using rapid field assessment and remote sensing', *Journal of Environmental Management*, 97, pp. 102-108. doi: 10.1016/j.jenvman.2011.12.005

Řehounková, K. and Prach, K. (2006) 'Spontaneous vegetation succession in disused gravel-sand pits: role of local site and landscape factors', *Journal of Vegetation Science*, 17(5), pp. 583-590. doi: 10.1111/j.1654-1103.2006.tb02482.x

(2008) 'Spontaneous vegetation succession in gravel-sand pits: a potential for restoration', *Restoration Ecology*, 16(2), pp. 305-312. doi: 10.1111/j.1526-100X.2007.00316.x

Remke, E., Brouwer, E., Kooijman, A., Blindow, I., Esselink, H. and Roelofs, J.G.M. (2009) 'Even low to medium nitrogen deposition impacts vegetation of dry, coastal dunes around the Baltic Sea', *Environmental Pollution*, 157(3), pp. 792-800. doi: 10.1016/j.envpol.2008.11.020

Remke, E.S. and Blindow, I. (2011) 'Site specific factors have an overriding impact on Baltic dune vegetation change under low to moderate N-deposition -

a case study from Hiddensee island', *Journal of Coastal Conservation*, 15(1), pp. 87-97. doi: 10.1007/s11852-010-0123-2

Ribera, I., Dolédec, S., Downie, I.S. and Foster, G.N. (2001) 'Effect of land disturbance and stress on species traits of ground beetle assemblages', *Ecology*, 82(4), pp. 1112-1129. doi: 10.1890/0012-9658(2001)082[1112:EOLDAS]2.0.CO;2

Ribera, I., Foster, G.N., Downie, I.S., McCracken, D.I. and Abernethy, V.J. (1999) 'A comparative study of the morphology and life traits of Scottish ground beetles (Coleoptera, Carabidae)', Annales Zoologici Fennici, 36(1), pp. 21-37. Available at: http://www.jstor.org/stable/23735639 (Accessed: 8 October 2017).

Richards, O.W. (1980) *Scolioidea, Vespoidea and Sphecoidea: Hymenoptera, Aculeata.* London: Handbooks for the Identification of British Insects Vol. VI, Part 3(b).

Ricketts, T.H. (2001) 'The matrix matters: effective isolation in fragmented landscapes', *The American Naturalist*, 158(1), pp. 87-99. doi: 10.1086/320863

Ridding, L.E., Redhead, J.W. and Pywell, R.F. (2015) 'Fate of semi-natural grassland in England between 1960 and 2013: a test of national conservation policy', *Global Ecology and Conservation*, 4, pp. 516-525. doi: 10.1016/j.gecco.2015.10.004

Roberts, M.J. (1985) *The Spiders of Great Britain and Ireland: Volume 1: Atypidae to Theridiosomatidae.* Colchester: Harley Books.

Roberts, M.J. (1987) *The Spiders of Great Britain and Ireland: Volume 2: Linyphiidae and check list.* Colchester: Harley Books.

Robins, J. (2013) *Baseline habitat assessment and management plan specification for the Buckingham Hill former landfill site*. Unpublished report.

Robins, J., Henshall, S. and Farr, A. (2013) *The state of brownfields in the Thames Gateway*. Available at: https://www.buglife.org.uk/sites/default/files/The%20State%20of%20Brownfields%20in%20the%20Thames%20Gateway_0_0.pdf (Accessed: 9 October 2017).

Robinson, S.L. and Lundholm, J.T. (2012) 'Ecosystem services provided by urban spontaneous vegetation', *Urban Ecosystems*, 15(3), pp. 545-557. doi: 10.1007/s11252-012-0225-8

Roff, D.A. (1986) 'The evolution of wing dimorphism in insects', *Evolution*, 40(5), pp. 1009-1020. doi: 10.1111/j.1558-5646.1986.tb00568.x

Rogers, M.E. and Potter, D.A. (2003) 'Effects of spring imidacloprid application for white grub control on parasitism of Japanese beetle (Coleoptera: Scarabaeidae) by *Tiphia vernalis* (Hymenoptera: Tiphiidae)', *Journal of*

Economic Entomology, 96(5), pp. 1412-1419. doi: 10.1603/0022-0493-96.5.1412

Rognes, K. (1991) *Blowflies (Diptera, Calliphoridae) of Fennoscandia and Denmark.* Leiden: E.J. Brill. Fauna Entomologica Scandinavica Volume 24.

Rossell, I.M., Moorhead, K.K., Alvarado, H. and Warren, R.J. (2009) 'Succession of a Southern Appalachian mountain wetland six years following hydrologic and microtopographic restoration', *Restoration Ecology*, 17(2), pp. 205-214. doi: 10.1111/j.1526-100X.2008.00372.x

Rossetti, I., Bagella, S., Cappai, C., Caria, M.C., Lai, R., Roggero, P.P., da Silva, P.M., Sousa, J.P., Querner, P. and Seddaiu, G. (2015) 'Isolated cork oak trees affect soil properties and biodiversity in a Mediterranean wooded grassland', *Agriculture, Ecosystems & Environment*, 202, pp. 203-216. doi: 10.1016/j.agee.2015.01.008

RoTAP (2012) *Review of transboundary air pollution (RoTAP): acidification, eutrophication, ground level ozone and heavy metals in the UK*. Available at: http://www.rotap.ceh.ac.uk/files/CEH%20RoTAP_0.pdf (Accessed: 9 October 2017).

Royal Society for the Protection of Birds, The Land Trust and Buglife (2012) *Canvey Wick Management Plan: 2012-2017*. Unpublished.

Rozkošný, R., Gregor, F. and Pont, A.C. (1997) *The European Fanniidae (Diptera).* Acta scientiarum naturalium Academiae scientiarum Bohemicae - Brno, v.31(2), Brno, Czech Republic: Institute of Landscape Ecology.

Rumble, H. and Gange, A.C. (2013) 'Soil microarthropod community dynamics in extensive green roofs', *Ecological Engineering*, 57, pp. 197-204. doi: 10.1016/j.ecoleng.2013.04.012

Rupprecht, C.D.D., Byrne, J.A. and Lo, A.Y. (2016) 'Memories of vacant lots: how and why residents used informal urban green space as children and teenagers in Brisbane, Australia, and Sapporo, Japan', *Children's Geographies*, 14(3), pp. 340-355. doi: 10.1080/14733285.2015.1048427

Rupprecht, C.D.D., Byrne, J.A., Ueda, H. and Lo, A.Y. (2015) ''It's real, not fake like a park': residents' perception and use of informal urban green-space in Brisbane, Australia and Sapporo, Japan', *Landscape and Urban Planning*, 143, pp. 205-218. doi: 10.1016/j.landurbplan.2015.07.003

Rusch, A., Birkhofer, K., Bommarco, R., Smith, H.G. and Ekbom, B. (2015) 'Predator body sizes and habitat preferences predict predation rates in an agroecosystem', *Basic and Applied Ecology*, 16(3), pp. 250-259. doi: 10.1016/j.baae.2015.02.003

Rushton, S.P. (1988) 'The effects of scrub management regimes on the spider fauna of chalk grassland, Castor Hanglands National Nature Reserve, Cambridgeshire, UK', *Biological Conservation,* 46(3), pp. 169-182.

Rushton, S.P., Eyre, M.D. and Luff, M.L. (1990) 'The effects of scrub management on the ground beetles of oolitic limestone grassland at Castor Hanglands National Nature Reserve, Cambridgeshire, UK', *Biological Conservation,* 51(2), pp. 97-111.

Rushton, S.P., Luff, M.L. and Eyre, M.D. (1989) 'Effects of pasture improvement and management on the ground beetle and spider communities of upland grasslands', *Journal of Applied Ecology*, 26(2), pp. 489-503. doi: 10.2307/2404076

Sadler, J., Bates, A., Donovan, R. and Bodnar, S. (2011) 'Building for biodiversity: accommodating people and wildlife in cities', in Niemelä, J., Breuste, J.H., Guntenspergen, G., McIntyre, N.E., Elmqvist, T. & James, P. (eds.) *Urban ecology: patterns, processes and applications*. Oxford: Oxford University Press, pp. 286-297.

Saila, S.B., Pratt, S.D. and Polgar, T.T. (1972) *Dredge spoil disposal in Rhode Island Sound*. Available at: nsgl.gso.uri.edu/riu/riut72002.pdf (Accessed: 9 October 2017).

Samu, F. and Szinetár, C. (2002) 'On the nature of agrobiont spiders', *Journal of Arachnology*, 30(2), pp. 389-402. doi: 10.1636/0161-8202(2002)030[0389:OTNOAS]2.0.CO;2

Sarthou, J.-P., Ouin, A., Arrignon, F., Barreau, G. and Bouyjou, B. (2005) 'Landscape parameters explain the distribution and abundance of *Episyrphus balteatus* (Diptera: Syrphidae)', *European Journal of Entomology*, 102(3), pp. 539-545. doi: 10.14411/eje.2005.077

Saunders, D.A., Hobbs, R.J. and Margules, C.R. (1991) 'Biological consequences of ecosystem fragmentation: a review', *Conservation Biology*, 5(1), pp. 18-32. doi: 10.1111/j.1523-1739.1991.tb00384.x

Saure, C. (1996) 'Urban habitats for bees: the example of the city of Berlin', in Matheson, A., Buchmann, S.L., O'Toole, C., Westrich, P. & Williams, I.H. (eds.) *The conservation of bees*. London: Academic Press, pp. 47-54.

Schadek, U., Strauss, B., Biedermann, R. and Kleyer, M. (2009) 'Plant species richness, vegetation structure and soil resources of urban brownfield sites linked to successional age', *Urban Ecosystems*, 12(2), pp. 115-126. doi: 10.1007/s11252-008-0072-9

Schirmel, J. (2010) 'Short-term effects of modern heathland management measures on carabid beetles (Coleoptera: Carabidae)', *Applied Ecology and Environmental Research,* 8(3), pp. 165-175.

Schirmel, J., Blindow, I. and Buchholz, S. (2012) 'Life-history trait and functional diversity patterns of ground beetles and spiders along a coastal heathland successional gradient', *Basic and Applied Ecology*, 13(7), pp. 606-614. doi: 10.1016/j.baae.2012.08.015

Schirmel, J. and Buchholz, S. (2011) 'Response of carabid beetles (Coleoptera: Carabidae) and spiders (Araneae) to coastal heathland succession', *Biodiversity and Conservation*, 20(7), pp. 1469-1482. doi: 10.1007/s10531-011-0038-8

Schirmel, J., Mantilla-Contreras, J., Blindow, I. and Fartmann, T. (2011) 'Impacts of succession and grass encroachment on heathland Orthoptera', *Journal of Insect Conservation*, 15(5), pp. 633-642. doi: 10.1007/s10841-010-9362-7

Schirmel, J., Mantilla-Contreras, J., Gauger, D. and Blindow, I. (2015) 'Carabid beetles as indicators for shrub encroachment in dry grasslands', *Ecological Indicators*, 49, pp. 76-82. doi: 10.1016/j.ecolind.2014.09.041

Schirmel, J., Thiele, J., Entling, M.H. and Buchholz, S. (2016) 'Trait composition and functional diversity of spiders and carabids in linear landscape elements', *Agriculture, Ecosystems & Environment*, 235, pp. 318-328. doi: 10.1016/j.agee.2016.10.028

Schlacher, T.A. and Morrison, J.M. (2008) 'Beach disturbance caused by off-road vehicles (ORVs) on sandy shores: relationship with traffic volumes and a new method to quantify impacts using image-based data acquisition and analysis', *Marine Pollution Bulletin*, 56(9), pp. 1646-1649. doi: 10.1016/j.marpolbul.2008.06.008

Schmidt, M.H., Lefebvre, G., Poulin, B. and Tscharntke, T. (2005) 'Reed cutting affects arthropod communities, potentially reducing food for passerine birds', *Biological Conservation*, 121(2), pp. 157-166. doi: 10.1016/j.biocon.2004.03.032

Schmidt, M.H., Thies, C., Nentwig, W. and Tscharntke, T. (2008) 'Contrasting responses of arable spiders to the landscape matrix at different spatial scales', *Journal of Biogeography*, 35(1), pp. 157-166. doi: 10.1111/j.1365-2699.2007.01774.x

Schmidt, M.H. and Tscharntke, T. (2005) 'The role of perennial habitats for Central European farmland spiders', *Agriculture, Ecosystems & Environment*, 105(1), pp. 235-242. doi: 10.1016/j.agee.2004.03.009

Schüepp, C., Herrmann, J.D., Herzog, F. and Schmidt-Entling, M.H. (2011) 'Differential effects of habitat isolation and landscape composition on wasps, bees, and their enemies', *Oecologia*, 165(3), pp. 713-721. doi: 10.1007/s00442-010-1746-6

Schuldt, A., Bruelheide, H., Durka, W., Michalski, S.G., Purschke, O. and Assmann, T. (2014) 'Tree diversity promotes functional dissimilarity and

maintains functional richness despite species loss in predator assemblages', *Oecologia*, 174(2), pp. 533-543. doi: 10.1007/s00442-013-2790-9

Schwarz, N., Moretti, M., Bugalho, M.N., Davies, Z.G., Haase, D., Hack, J., Hof, A., Melero, Y., Pett, T.J. and Knapp, S. (2017) 'Understanding biodiversity-ecosystem service relationships in urban areas: A comprehensive literature review', *Ecosystem Services*, 27(Part A), pp. 161-171. doi: https://doi.org/10.1016/j.ecoser.2017.08.014

Schwerk, A. (2014) 'Changes in carabid beetle fauna (Coleoptera: Carabidae) along successional gradients in post-industrial areas in central Poland', *European Journal of Entomology*, 111(5), pp. 1-9. doi: 10.14411/eje.2014.076

Scott, A.G., Oxford, G.S. and Selden, P.A. (2006) 'Epigeic spiders as ecological indicators of conservation value for peat bogs', *Biological Conservation*, 127(4), pp. 420-428. doi: 10.1016/j.biocon.2005.09.001

Secretariat of the Convention on Biological Diversity (2014) *Global Biodiversity Outlook 4: a mid-term assessment of progress towards the implementation of the Strategic Plan for Biodiversity 2011-2020*. Available at: https://www.cbd.int/gbo4/ (Accessed: 9 October 2017).

Shaw, P. (2003) 'Collembola of pulverised fuel ash sites in east London', *European Journal of Soil Biology*, 39(1), pp. 1-8. doi: 10.1016/S1164-5563(02)00002-X

(2009a) 'Soil and fertilizer amendments and edge effects on the floral succession of pulverized fuel ash', *Restoration Ecology*, 17(1), pp. 68-77. doi: 10.1111/j.1526-100X.2007.00345.x

Shaw, P.J.A. (1992) 'A preliminary study of successional changes in vegetation and soil development on unamended fly ash (PFA) in southern England', *Journal of Applied Ecology*, 29(3), pp. 728-736. doi: 10.2307/2404482

Shaw, P.J.A. (1996) 'Role of seedbank substrates in the revegetation of fly ash and gypsum in the United Kingdom', *Restoration Ecology*, 4(1), pp. 61-70. doi: 10.1111/j.1526-100X.1996.tb00108.x

Shaw, P.J.A. (1998) 'Conservation management of industrial wastes', *Journal of Practical Ecology and Conservation,* 2(1), pp. 13-18.

(2009b) 'Succession on the PFA/gypsum trial mounds at Drax power station: the first fifteen years', *Journal of Practical Ecology and Conservation,* 8(2), pp. 7-19.

(2011) 'Management of brownfield sites for biodiversity', *Aspects of Applied Biology,* (108), pp. 179-192.

Sheffield, C.S., Wilkes, M.A., Christopher Cutler, G. and Hermanutz, L. (2015) 'An artificial nesting substrate for Osmia species that nest under stones, with focus on *Osmia inermis* (Hymenoptera: Megachilidae)', *Insect Conservation and Diversity*, 8(2), pp. 189-192. doi: 10.1111/icad.12095

Shoard, M. (2002) 'Edgelands', in Jenkins, J. (ed.) *Remaking the landscape: the changing face of Britain*. London: Profile Books, pp. 117-146.

Shokralla, S., Porter, T.M., Gibson, J.F., Dobosz, R., Janzen, D.H., Hallwachs, W., Golding, G.B. and Hajibabaei, M. (2015) 'Massively parallel multiplex DNA sequencing for specimen identification using an Illumina MiSeq platform', *Scientific reports*, 5, 9687 (7pp). doi: 10.1038/srep09687

Simberloff, D. and Abele, L.G. (1982) 'Refuge design and island biogeographic theory: effects of fragmentation', *The American Naturalist*, 120(1), pp. 41-50. doi: 10.1086/283968

Simberloff, D.S. and Wilson, E.O. (1969) 'Experimental zoogeography of islands: the colonization of empty islands', *Ecology*, 50(2), pp. 278-296. doi: 10.2307/1934856

Simmons, M.E., Ben Wu, X. and Whisenant, S.G. (2011) 'Plant and soil responses to created microtopography and soil treatments in bottomland hardwood forest restoration', *Restoration Ecology*, 19(1), pp. 136-146. doi: 10.1111/j.1526-100X.2009.00524.x

Simons, N.K., Weisser, W.W. and Gossner, M.M. (2016) 'Multi-taxa approach shows consistent shifts in arthropod functional traits along grassland land-use intensity gradient', *Ecology*, 97(3), pp. 754-764. doi: 10.1890/15-0616.1

Sinnett, D., Carmichael, L., Williams, K. and Miner, P. (2014) *From wasted space to living spaces: the availability of brownfield land for housing development in England*. Available at: http://www.cpre.org.uk/resources/housing-and-planning/housing/item/download/3847 (Accessed: 9 October 2017).

Skidmore, P. (1985) *The biology of the Muscidae of the world.* Dordrecht, The Netherlands: Dr. W. Junk Publishers.

Skilbeck, C.A. and Anderson, M. (1996) 'The ultrastructure, morphology and distribution of sensilla on the antennae of the adult parasitoids *Aleochara bilineata* Gyll. and *Aleochara bipustulata* L. (Coleoptera: Staphylinidae)', *International Journal of Insect Morphology and Embryology*, 25(3), pp. 261-280. doi: 10.1016/0020-7322(96)00005-0

Skvarla, M.J., Larson, J.L. and Dowling, A.P.G. (2014) 'Pitfalls and preservatives: a review', *Journal of the Entomological Society of Ontario,* 145, pp. 15-43.

Small, E., Sadler, J.P. and Telfer, M. (2006) 'Do landscape factors affect brownfield carabid assemblages?', *Science of the Total Environment*, 360(1), pp. 205-222. doi: 10.1016/j.scitotenv.2005.08.051

Small, E.C., Sadler, J.P. and Telfer, M.G. (2003) 'Carabid beetle assemblages on urban derelict sites in Birmingham, UK', *Journal of Insect Conservation*, 6(4), pp. 233-246. doi: 10.1023/A:1024491111572

Smith, R.K., Dicks, L.V., Mitchell, R. and Sutherland, W.J. (2014) 'Comparative effectiveness research: the missing link in conservation', Conservation Evidence, 11, pp. 2-6. Available at: https://www.conservationevidence.com/individual-study/5475 (Accessed: 8 October 2017).

Sobek, S., Tscharntke, T., Scherber, C., Schiele, S. and Steffan-Dewenter, I. (2009) 'Canopy vs. understory: does tree diversity affect bee and wasp communities and their natural enemies across forest strata?', *Forest Ecology and Management*, 258(5), pp. 609-615. doi: 10.1016/j.foreco.2009.04.026

Sousa, W.P. (1984) 'The role of disturbance in natural communities', *Annual Review of Ecology and Systematics*, 15(1), pp. 353-391. doi: 10.1146/annurev.es.15.110184.002033

Southwood, T.R.E. (1977) 'Habitat, the templet for ecological strategies?', *Journal of Animal Ecology*, 46(2), pp. 337-365. doi: 10.2307/3817

(1988) 'Tactics, strategies and templets', *Oikos*, 52(1), pp. 3-18. doi: 10.2307/3565974

Spake, R., Barsoum, N., Newton, A.C. and Doncaster, C.P. (2016) 'Drivers of the composition and diversity of carabid functional traits in UK coniferous plantations', *Forest Ecology and Management*, 359, pp. 300-308. doi: 10.1016/j.foreco.2015.10.008

Spence, J.R. and Niemelä, J.K. (1994) 'Sampling carabid assemblages with pitfall traps: the madness and the method', *The Canadian Entomologist*, 126(3), pp. 881-894. doi: 10.4039/Ent126881-3

Srba, M. and Heneberg, P. (2012) 'Nesting habitat segregation between closely related terricolous sphecid species (Hymenoptera: Spheciformes): key role of soil physical characteristics', *Journal of Insect Conservation*, 16(4), pp. 557-570. doi: 10.1007/s10841-011-9441-4

Ssymank, A., Kearns, C.A., Pape, T. and Thompson, F.C. (2008) 'Pollinating flies (Diptera): a major contribution to plant diversity and agricultural production', *Biodiversity*, 9(1-2), pp. 86-89. doi: 10.1080/14888386.2008.9712892

Stace, C. (2010) *New flora of the British Isles.* 3rd edn. Cambridge: Cambridge University Press.

Staley, J., Ball, S., Ridding, L. and Pywell, R. (2015) *Final report for LM0202: scoping study to develop a methodology that can be used to assess the value of agri-environment scheme options in the creation and maintenance of small-scale habitat mosaics to benefit priority species*. Available at: http://randd.defra.gov.uk/Document.aspx?Document=13627_LM0202_ScopingStudy_FinalReport.pdf (Accessed: 9 October 2017).

Staniec, B. (1999) 'Comparative morphology of the mature larvae of the *Bledius* species inhabiting the loessic soils of the middle-eastern Poland', *Deutsche Entomologische Zeitschrift*, 46(1), pp. 43-74. doi: 10.1002/mmnd.19990460103

Steel, W.O. (1949) '*Planeustomus palpalis* (Erichs.) and *Scopaeus sulcicollis* (Steph.) (Col., Staphylinidae) in a garden in Middlesex', *Entomologist's Monthly Magazine,* 105, p. 50.

Steffan-Dewenter, I., Münzenberg, U., Bürger, C., Thies, C. and Tscharntke, T. (2002) 'Scale-dependent effects of landscape context on three pollinator guilds', *Ecology*, 83(5), pp. 1421-1432. doi: 10.1890/0012-9658(2002)083[1421:SDEOLC]2.0.CO;2

Steffan-Dewenter, I. and Schiele, S. (2004) 'Nest-site fidelity, body weight and population size of the red mason bee, Osmia rufa (Hymenoptera: Megachilidae), evaluated by mark-recapture experiments', *Entomologia Generalis*, 27(2), pp. 123-131. doi: 10.1127/entom.gen/27/2004/123

Steffan-Dewenter, I. and Tscharntke, T. (2000) 'Butterfly community structure in fragmented habitats', *Ecology Letters*, 3(5), pp. 449-456. doi: 10.1111/j.1461-0248.2000.00175.x

Sterling, P.H., Gibson, C.W.D. and Brown, V.K. (1992) 'Leaf miner assemblies: effects of plant succession and grazing management', *Ecological Entomology*, 17(2), pp. 167-178. doi: 10.1111/j.1365-2311.1992.tb01175.x

Stevens, C.J., Payne, R.J., Kimberley, A. and Smart, S.M. (2016) 'How will the semi-natural vegetation of the UK have changed by 2030 given likely changes in nitrogen deposition?', *Environmental Pollution*, 208(Part B), pp. 879-889. doi: 10.1016/j.envpol.2015.09.013

Stewart, K.E.J., Bourn, N.A.D. and Thomas, J.A. (2001) 'An evaluation of three quick methods commonly used to assess sward height in ecology', *Journal of Applied Ecology*, 38(5), pp. 1148-1154. doi: 10.1046/j.1365-2664.2001.00658.x

Stoeffler, M., Tolasch, T. and Steidle, J.L.M. (2011) 'Three beetles - three concepts. Different defensive strategies of congeneric myrmecophilous beetles', *Behavioral Ecology and Sociobiology*, 65(8), pp. 1605-1613. doi: 10.1007/s00265-011-1171-9

Štokmane, M., Spuņģis, V. and Cera, I. 'Spider (Arachnida: Araneae) species richness, community structure and ecological factors influencing spider diversity

in the calcareous fens of Latvia', *The 54th International Scientific Conference of Daugavpils University*. Daugavpils Universitāte, Latvia, 21 January 2013. Latvia: Daugavpils University, pp. 45-55. Available at: https://www.researchgate.net/profile/Maija_Stokmane2/publication/271586991_Spider_Arachnida_Araneae_species_richness_community_structure_and_ecological_factors_influencing_spider_diversity_in_the_calcareous_fens_of_Latvia/links/54cd15260cf24601c08cb291/Spider-Arachnida-Araneae-species-richness-community-structure-and-ecological-factors-influencing-spider-diversity-in-the-calcareous-fens-of-Latvia.pdf (Accessed: 11 October 2017).

Stone, G.N. (1994) 'Activity patterns of females of the solitary bee *Anthophora plumipes* in relation to temperature, nectar supplies and body size', *Ecological Entomology*, 19(2), pp. 177-189. doi: 10.1111/j.1365-2311.1994.tb00408.x

Stovin, V. (2010) 'The potential of green roofs to manage urban stormwater', *Water and Environment Journal*, 24(3), pp. 192-199. doi: 10.1111/j.1747-6593.2009.00174.x

Strauss, B. and Biedermann, R. (2006) 'Urban brownfields as temporary habitats: driving forces for the diversity of phytophagous insects', *Ecography*, 29(6), pp. 928-940. doi: 10.1111/j.2006.0906-7590.04716.x

Strauss, B. and Biedermann, R. (2008) 'Fit for succession - community structure and life strategies of leafhoppers in urban brownfields', *Ecological Entomology*, 33, pp. 107-118. doi: 10.1111/j.1365-2311.2007.00944.x

Strickler, K., Scott, V.L. and Fischer, R.L. (1996) 'Comparative nesting ecology of two sympatric leafcutting bees that differ in body size (Hymenoptera: Megachilidae)', Journal of the Kansas Entomological Society, 69(1), pp. 26-44. Available at: http://www.jstor.org/stable/25085646 (Accessed: 8 October 2017).

Strohm, E. (2000) 'Factors affecting body size and fat content in a digger wasp', *Oecologia*, 123(2), pp. 184-191. doi: 10.1007/s004420051004

Sunderland, K.D. (1975) 'The diet of some predatory arthropods in cereal crops', *Journal of Applied Ecology*, 12(2), pp. 507-515. doi: 10.2307/2402171

Szefer, P., Carmona, C.P., Chmel, K., Konečná, M., Libra, M., Molem, K., Novotný, V., Segar, S.T., Švamberková, E. and Topliceanu, T.S. (2017) 'Determinants of litter decomposition rates in a tropical forest: functional traits, phylogeny and ecological succession', *Oikos*, 126(8), pp. 1101-1111. doi: 10.1111/oik.03670

Talarico, F., Giglio, A., Pizzolotto, R. and Brandmayr, P. (2016) 'A synthesis of feeding habits and reproduction rhythm in Italian seed-feeding ground beetles (Coleoptera: Carabidae)', *European Journal of Entomology*, 113, pp. 325-336. doi: 10.14411/eje.2016.042

Tang, M., Hardman, C.J., Ji, Y., Meng, G., Liu, S., Tan, M., Yang, S., Moss, E.D., Wang, J. and Yang, C. (2015) 'High-throughput monitoring of wild bee diversity and abundance via mitogenomics', *Methods in Ecology and Evolution*, 6(9), pp. 1034-1043. doi: 10.1111/2041-210X.12416

Taranath, B.S. (2016) *Structural analysis and design of tall buildings: steel and composite construction.* Boca Raton, FL: CRC Press.

Tarrant, S., Ollerton, J., Rahman, M.L., Tarrant, J. and McCollin, D. (2013) 'Grassland restoration on landfill sites in the East Midlands, United Kingdom: an evaluation of floral resources and pollinating insects', *Restoration Ecology,* 21(5), pp. 560-568.

Taudiere, A. and Violle, C. (2015) cati: an R package using functional traits to detect and quantify multi-level community assembly processes (Version 0.99.1) [R package]. Available at: https://cran.r-project.org/web/packages/cati/index.html (Accessed: 7 October 2017).

Telfer, M.G. (2016) *A review of the beetles of Great Britain: ground beetles (Carabidae): species status No. 25*. Available at: http://publications.naturalengland.org.uk/publication/6270849377107968 (Accessed: 9 October 2017).

Telnov, D. (2010) 'Ant-like flower beetles (Coleoptera: Anthicidae) of the UK, Ireland and Channel Isles', *British Journal of Entomology and Natural History,* 23(2), pp. 99-120.

ten Harkel, M.J. and van der Meulen, F. (1996) 'Impact of grazing and atmospheric nitrogen deposition on the vegetation of dry coastal dune grasslands', *Journal of Vegetation Science*, 7(3), pp. 445-452. doi: 10.2307/3236288

ter Braak, C.J.F., Cormont, A. and Dray, S. (2012) 'Improved testing of species traits–environment relationships in the fourth-corner problem', *Ecology*, 93(7), pp. 1525-1526. doi: 10.1890/12-0126.1

ter Braak, C.J.F. and Šmilauer, P. (2014) CANOCO (Version 5.04) [Computer program]. Microcomputer Power, Ithaca, NY.

ter Braak, C.J.F. and Verdonschot, P.F.M. (1995) 'Canonical correspondence analysis and related multivariate methods in aquatic ecology', *Aquatic Sciences,* 57(3), pp. 255-289.

Tewksbury, J.J., Levey, D.J., Haddad, N.M., Sargent, S., Orrock, J.L., Weldon, A., Danielson, B.J., Brinkerhoff, J., Damschen, E.I. and Townsend, P. (2002) 'Corridors affect plants, animals, and their interactions in fragmented landscapes', *Proceedings of the National Academy of Sciences*, 99(20), pp. 12923-12926. doi: 10.1073/pnas.202242699

The Economics of Ecosystems and Biodiversity (2010) *The economics of ecosystems and biodiversity: ecological and economic foundations.* London: Earthscan.

The Scottish Government (Scotland) (2008) *Scottish Planning Policy 3: Planning for Homes*. Available at: http://www.gov.scot/Resource/Doc/233260/0063937.pdf (Accessed: 9 October 2017).

Thomas, A.S. (1963) 'Further changes in vegetation since the advent of myxomatosis', *Journal of Ecology*, 51(1), pp. 151-186. doi: 10.2307/2257511

Thomas, C.D. and Harrison, S. (1992) 'Spatial dynamics of a patchily distributed butterfly species', *Journal of Animal Ecology*, 61(2), pp. 437-446. doi: 10.2307/5334

Thomas, C.F.G., Brown, N.J. and Kendall, D.A. (2006) 'Carabid movement and vegetation density: implications for interpreting pitfall trap data from split-field trials', *Agriculture, Ecosystems & Environment*, 113(1–4), pp. 51-61. doi: 10.1016/j.agee.2005.08.033

Thomas, C.F.G., Parkinson, L. and Marshall, E.J.P. (1998) 'Isolating the components of activity-density for the carabid beetle *Pterostichus melanarius* in farmland', *Oecologia*, 116(1-2), pp. 103-112. doi: 10.1007/s004420050568

Thomas, J.A., Telfer, M.G., Roy, D.B., Preston, C.D., Greenwood, J.J.D., Asher, J., Fox, R., Clarke, R.T. and Lawton, J.H. (2004) 'Comparative losses of British butterflies, birds, and plants and the global extinction crisis', *Science*, 303(5665), pp. 1879-1881. doi: 10.1126/science.1095046

Thomas, J.A., Thomas, C.D., Simcox, D.J. and Clarke, R.T. (1986) 'Ecology and declining status of the silver-spotted skipper butterfly (*Hesperia comma*) in Britain', *Journal of Applied Ecology*, 23(2), pp. 365-380. doi: 10.2307/2404023

Thomas, M.B., Wratten, S.D. and Sotherton, N.W. (1991) 'Creation of 'island' habitats in farmland to manipulate populations of beneficial arthropods: predator densities and emigration', *Journal of Applied Ecology*, 28(3), pp. 906-917. doi: 10.2307/2404216

Thomas, S.R., Goulson, D. and Holland, J.M. (2001) 'Resource provision for farmland gamebirds: the value of beetle banks', *Annals of Applied Biology*, 139(1), pp. 111-118. doi: 10.1111/j.1744-7348.2001.tb00135.x

Thompson, K., Austin, K.C., Smith, R.M., Warren, P.H., Angold, P.G. and Gaston, K.J. (2003) 'Urban domestic gardens (I): putting small-scale plant diversity in context', *Journal of Vegetation Science*, 14(1), pp. 71-78. doi: 10.1111/j.1654-1103.2003.tb02129.x

Thompson, R.T. (1958) *Coleoptera: Phalacridae.* London: Royal Entomological Society of London. Handbooks for the Identification of British Insects Vol. V, Part 5(b).

Thuiller, W., Lavorel, S., Araújo, M.B., Sykes, M.T. and Prentice, I.C. (2005) 'Climate change threats to plant diversity in Europe', *Proceedings of the National Academy of Sciences of the United States of America*, 102(23), pp. 8245-8250. doi: 10.1073/pnas.0409902102

Tichanek, F. and Tropek, R. (2015) 'Conservation value of post-mining headwaters: drainage channels at a lignite spoil heap harbour threatened stream dragonflies', *Journal of Insect Conservation*, 19(5), pp. 975-985. doi: 10.1007/s10841-015-9814-1

Tonietto, R., Fant, J., Ascher, J., Ellis, K. and Larkin, D. (2011) 'A comparison of bee communities of Chicago green roofs, parks and prairies', *Landscape and Urban Planning*, 103(1), pp. 102-108. doi: 10.1016/j.landurbplan.2011.07.004

Topp, W., Simon, M., Kautz, G., Dworschak, U., Nicolini, F. and Prückner, S. (2001) 'Soil fauna of a reclaimed lignite open-cast mine of the Rhineland: improvement of soil quality by surface pattern', *Ecological Engineering*, 17(2), pp. 307-322. doi: 10.1016/S0925-8574(00)00147-6

Topp, W., Thelen, K. and Kappes, H. (2010) 'Soil dumping techniques and afforestation drive ground-dwelling beetle assemblages in a 25-year-old open-cast mining reclamation area', *Ecological Engineering*, 36(6), pp. 751-756. doi: 10.1016/j.ecoleng.2009.12.011

Topping, C.J. and Sunderland, K. (1992) 'Limitations to the use of pitfall traps in ecological studies exemplified by a study of spiders in a field of winter wheat', *Journal of Applied Ecology*, 29(2), pp. 485-491. doi: 10.2307/2404516

Townsend, C.R. and Hildrew, A.G. (1994) 'Species traits in relation to a habitat templet for river systems', *Freshwater Biology*, 31(3), pp. 265-275. doi: 10.1111/j.1365-2427.1994.tb01740.x

Traut, B.H. (2005) 'The role of coastal ecotones: a case study of the salt marsh/upland transition zone in California', *Journal of Ecology*, 93(2), pp. 279-290. doi: 10.1111/j.1365-2745.2005.00969.x

Tropek, R., Cerna, I., Straka, J., Cizek, O. and Konvicka, M. (2013a) 'Is coal combustion the last chance for vanishing insects of inland drift sand dunes in Europe?', *Biological Conservation*, 162, pp. 60-64. doi: 10.1016/j.biocon.2013.03.027

Tropek, R., Cerna, I., Straka, J., Kadlec, T., Pech, P., Tichanek, F. and Sebek, P. (2014) 'Restoration management of fly ash deposits crucially influence their conservation potential for terrestrial arthropods', *Ecological Engineering*, 73, pp. 45-52. doi: 10.1016/j.ecoleng.2014.09.011

Tropek, R., Cerna, I., Straka, J., Kocarek, P., Malenovsky, I., Tichanek, F. and Sebek, P. (2016) 'In search for a compromise between biodiversity conservation and human health protection in restoration of fly ash deposits: effect of anti-dust treatments on five groups of arthropods', *Environmental Science and Pollution Research*, 23(14), pp. 13653-13660. doi: 10.1007/s11356-015-4382-1

Tropek, R., Hejda, M., Kadlec, T. and Spitzer, L. (2013b) 'Local and landscape factors affecting communities of plants and diurnal Lepidoptera in black coal spoil heaps: ikmplications for restoration management', *Ecological Engineering*, 57, pp. 252-260. doi: 10.1016/j.ecoleng.2013.04.024

Tropek, R., Kadlec, T., Hejda, M., Kocarek, P., Skuhrovec, J., Malenovsky, I., Vodka, S., Spitzer, L., Banar, P. and Konvicka, M. (2012) 'Technical reclamations are wasting the conservation potential of post-mining sites. A case study of black coal spoil dumps', *Ecological Engineering*, 43, pp. 13-18. doi: 10.1016/j.ecoleng.2011.10.010

Tropek, R., Kadlec, T., Karesova, P., Spitzer, L., Kocarek, P., Malenovsky, I., Banar, P., Tuf, I.H., Hejda, M. and Konvicka, M. (2010) 'Spontaneous succession in limestone quarries as an effective restoration tool for endangered arthropods and plants', *Journal of Applied Ecology*, 47(1), pp. 139-147. doi: 10.1111/j.1365-2664.2009.01746.x

Tropek, R. and Konvicka, M. (2008) 'Can quarries supplement rare xeric habitats in a piedmont region? Spiders of the Blansky les Mts, Czech Republic', *Land Degradation & Development*, 19(1), pp. 104-114. doi: 10.1002/ldr.817

Tropek, R., Spitzer, L. and Konvicka, M. (2008) 'Two groups of epigeic arthropods differ in colonising of piedmont quarries: the necessity of multi-taxa and life-history traits approaches in the monitoring studies', *Community Ecology*, 9(2), pp. 177-184. doi: 10.1556/ComEc.9.2008.2.6

Tscharntke, T., Steffan-Dewenter, I., Kruess, A. and Thies, C. (2002) 'Contribution of small habitat fragments to conservation of insect communities of grassland-cropland landscapes', *Ecological Applications*, 12(2), pp. 354-363. doi: 10.1890/1051-0761(2002)012[0354:COSHFT]2.0.CO;2

Tucker, D., Ash, H. and Plant, C. (2005) *Review of the coverage of urban habitats and species within the UK Biodiversity Action Plan*. Available at: http://publications.naturalengland.org.uk/publication/91034 (Accessed: 9 October 2017).

Turin, H., Alders, K., den Boer, P.J., van Essen, S., Heijennan, T., Laane, W. and Penterman, E. (1991) 'Ecological characterization of carabid species (Coleoptera, Carabidae) in the Netherlands from thirty years of pitfall sampling', *Tijdschrift voor Entomologie,* 134, pp. 279-304.

Turin, H. and Den Boer, P.J. (1988) 'Changes in the distribution of carabid beetles in The Netherlands since 1880. II. Isolation of habitats and long-term time trends in the occurence of carabid species with different powers of

dispersal (Coleoptera, Carabidae)', *Biological Conservation*, 44(3), pp. 179-200. doi: 10.1016/0006-3207(88)90101-2

Turner, E.C., Granroth, H.M.V., Johnson, H.R., Lucas, C.B.H., Thompson, A.M., Froy, H., German, R.N. and Holdgate, R. (2009) 'Habitat preference and dispersal of the Duke of Burgundy butterfly (*Hamearis lucina*) on an abandoned chalk quarry in Bedfordshire, UK', *Journal of Insect Conservation*, 13(5), pp. 475-486. doi: 10.1007/s10841-008-9194-x

Twerd, L., Krzyżyński, M., Waldon-Rudzionek, B. and Olszewski, P. (2017) 'Can soda ash dumping grounds provide replacement habitats for digger wasps (Hymenoptera, Apoidea, Spheciformes)?', *PLOS ONE*, 12(4), e0175664 (17pp). doi: 10.1371/journal.pone.0175664

Uetz, G.W. (1979) 'The influence of variation in litter habitats on spider communities', *Oecologia*, 40(1), pp. 29-42. doi: 10.1007/BF00388808

Uetz, G.W., Halaj, J. and Cady, A.B. (1999) 'Guild structure of spiders in major crops', Journal of Arachnology, 27(1), pp. 270-280. Available at: http://www.jstor.org/stable/3705998 (Accessed: 8 October 2017).

Uetz, G.W. and Unzicker, J.D. (1975) 'Pitfall trapping in ecological studies of wandering spiders', Journal of Arachnology, 3(2), pp. 101-111. Available at: http://www.jstor.org/stable/3705290 (Accessed: 8 October 2017).

UK Research on The Eutrophication and Acidification of Terrestrial Ecosystems (2009) *Terrestrial umbrella - Effects of eutrophication and acidification on terrestrial ecosystems: Annual Report 2009*. Available at: http://ukreate.defra.gov.uk/PDFs/Defra_TU_report_collated_final_2009.pdf (Accessed: 9 October 2017).

Umbanhowar, C.E. (1992) 'Early patterns of revegetation of artificial earthen mounds in a northern mixed prairie', *Canadian Journal of Botany*, 70(1), pp. 145-150. doi: 10.1139/b92-020

Van Emden, F.I. (1954) *Diptera Cyclorrhapha: Calyptrata (I) Section (a). Tachinidae and Calliphoridae.* London: Royal Entomological Society of London. Handbooks for the Identification of British Insects Vol. X, Part 4(a).

Vanbergen, A.J., Woodcock, B.A., Koivula, M., Niemelä, J., Kotze, D.J., Bolger, T., Golden, V., Dubs, F., Boulanger, G. and Serrano, J. (2010) 'Trophic level modulates carabid beetle responses to habitat and landscape structure: a pan-European study', *Ecological Entomology*, 35(2), pp. 226-235. doi: 10.1111/j.1365-2311.2010.01175.x

Vandewalle, M., de Bello, F., Berg, M.P., Bolger, T., Dolédec, S., Dubs, F., Feld, C.K., Harrington, R., Harrison, P.A., Lavorel, S., da Silva, P.M., Moretti, M., Niemelä, J., Santos, P., Sattler, T., Sousa, J.P., Sykes, M.T., Vanbergen, A.J. and Woodcock, B.A. (2010) 'Functional traits as indicators of biodiversity

response to land use changes across ecosystems and organisms', *Biodiversity and Conservation*, 19(10), pp. 2921-2947. doi: 10.1007/s10531-010-9798-9

Veer, M.A.C. and Kooijman, A.M. (1997) 'Effects of grass-encroachment on vegetation and soil in Dutch dry dune grasslands', *Plant and Soil*, 192(1), pp. 119-128. doi: 10.1023/A:1004280300820

Velle, L.G. and Vandvik, V. (2014) 'Succession after prescribed burning in coastal Calluna heathlands along a 340-km latitudinal gradient', *Journal of Vegetation Science*, 25(2), pp. 546-558. doi: 10.1111/jvs.12100

Vergnes, A., Le Viol, I. and Clergeau, P. (2012) 'Green corridors in urban landscapes affect the arthropod communities of domestic gardens', *Biological Conservation*, 145(1), pp. 171-178. doi: 10.1016/j.biocon.2011.11.002

Verhoeven, K.J.F., Simonsen, K.L. and McIntyre, L.M. (2005) 'Implementing false discovery rate control: increasing your power', *Oikos*, 108(3), pp. 643-647. doi: 10.1111/j.0030-1299.2005.13727.x

Vermeulen, H.J.W. (1993) 'The composition of the carabid fauna on poor sandy road-side verges in relation to comparable open areas', *Biodiversity and Conservation*, 2(4), pp. 331-350. doi: 10.1007/bf00114038

Villéger, S., Mason, N.W.H. and Mouillot, D. (2008) 'New multidimensional functional diversity indices for a multifaceted framework in functional ecology', *Ecology*, 89(8), pp. 2290-2301. doi: 10.1890/07-1206.1

Violle, C., Navas, M.L., Vile, D., Kazakou, E., Fortunel, C., Hummel, I. and Garnier, E. (2007) 'Let the concept of trait be functional!', *Oikos*, 116(5), pp. 882-892. doi: 10.1111/j.0030-1299.2007.15559.x

Volkl, W., Zwolfer, H., Romstock-Volkl, M. and Schmelzer, C. (1993) 'Habitat management in calcareous grasslands: effects on the insect community developing in flower heads of Cynarea', *Journal of Applied Ecology*, 30(2), pp. 307-315. doi: 10.2307/2404632

Ward, J.H. (1963) 'Hierarchical grouping to optimize an objective function', *Journal of the American statistical association*, 58(301), pp. 236-244. doi: 10.1080/01621459.1963.10500845

Warren, M.S., Hill, J.K., Thomas, J.A., Asher, J., Fox, R., Huntley, B., Roy, D.B., Telfer, M.G., Jeffcoate, S. and Harding, P. (2001) 'Rapid responses of British butterflies to opposing forces of climate and habitat change', *Nature*, 414(6859), pp. 65-69. doi: 10.1038/35102054

Warren, S.D. and Büttner, R. (2014) 'Restoration of heterogeneous disturbance regimes for the preservation of endangered species', *Ecological Restoration*, 32(2), pp. 189-196. doi: 10.3368/er.32.2.189

Warren, S.D., Holbrook, S.W., Dale, D.A., Whelan, N.L., Elyn, M., Grimm, W. and Jentsch, A. (2007) 'Biodiversity and the heterogeneous disturbance regime on military training lands', *Restoration Ecology*, 15(4), pp. 606-612. doi: 10.1111/j.1526-100X.2007.00272.x

Waste (England and Wales) (Amendment) Regulations 2012 (SI 2012/1889). Available at: http://www.legislation.gov.uk/uksi/2012/1889/pdfs/uksi_20121889_en.pdf (Accessed: 7 October 2017).

Watt, J.D. and Thorne, D.J. (1965) 'Composition and pozzolanic properties of pulverised fuel ashes. I. Composition of fly ashes from some British power stations and properties of their component particles', *Journal of Applied Chemistry*, 15(12), pp. 585-594. doi: 10.1002/jctb.5010151207

Weaving, A.J.S. (1989) 'Habitat selection and nest construction behaviour in some Afrotropical species of *Ammophila* (Hymenoptera: Sphecidae)', *Journal of Natural History*, 23(4), pp. 847-871. doi: 10.1080/00222938900770451

Webb, J. and Mott, N. (2015) 'Beetle records from dune slacks at Braunton Burrows, North Devon VC 4', *The Coleopterist,* 24(2), pp. 80-86.

Wedin, D.A. and Tilman, D. (1990) 'Species effects on nitrogen cycling: a test with perennial grasses', *Oecologia*, 84(4), pp. 433-441. doi: 10.1007/BF00328157

Weigel, B.M., Wang, L., Rasmussen, P.W., Butcher, J.T., Stewart, P.M., Simon, T.P. and Wiley, M.J. (2003) 'Relative influence of variables at multiple spatial scales on stream macroinvertebrates in the Northern Lakes and Forest ecoregion, USA', *Freshwater Biology*, 48(8), pp. 1440-1461. doi: 10.1046/j.1365-2427.2003.01076.x

Weithoff, G. (2003) 'The concepts of ‘plant functional types’ and ‘functional diversity’ in lake phytoplankton - a new understanding of phytoplankton ecology?', *Freshwater Biology*, 48(9), pp. 1669-1675. doi: 10.1046/j.1365-2427.2003.01116.x

Welch, R.C. (1989) '*Brachida exigua* (Heer) (Coleoptera: Staphylinidae) rediscovered in Kent', *Entomologist's Gazette,* 40, pp. 223-226.

Westrich, P. (1996) 'Habitat requirements of central European bees and the problems of partial habitats', in Matheson, A., Buchmann, S.L., O'Toole, C., Westrich, P. & Williams, I.H. (eds.) *The conservation of bees*. London: Academic Press, pp. 1-16.

White, I.M. (1977) 'The larvae of some British species of *Gyrophaena* Mannerheim (Coleoptera: Staphylinidae) with notes on the taxonomy and biology of the genus', *Zoological Journal of the Linnean Society*, 60(4), pp. 297-317. doi: 10.1111/j.1096-3642.1977.tb00837.x

White, P.S. and Walker, J.L. (1997) 'Approximating nature's variation: selecting and using reference information in restoration ecology', *Restoration Ecology*, 5(4), pp. 338-349. doi: 10.1046/j.1526-100X.1997.00547.x

Williams, G. and Hall, M. (1987) 'The loss of coastal grazing marshes in South and East England, with special reference to East Essex, England', *Biological Conservation*, 39(4), pp. 243-253. doi: 10.1016/0006-3207(87)90127-3

Williams, N.S.G., Lundholm, J. and Scott MacIvor, J. (2014) 'FORUM: Do green roofs help urban biodiversity conservation?', *Journal of Applied Ecology*, 51(6), pp. 1643-1649. doi: 10.1111/1365-2664.12333

Wilson, J.D., Morris, A.J., Arroyo, B.E., Clark, S.C. and Bradbury, R.B. (1999) 'A review of the abundance and diversity of invertebrate and plant foods of granivorous birds in northern Europe in relation to agricultural change', *Agriculture, Ecosystems & Environment*, 75(1–2), pp. 13-30. doi: 10.1016/S0167-8809(99)00064-X

Windhager, S., Steiner, F., Simmons, M.T. and Heymann, D. (2010) 'Toward ecosystem services as a basis for design', *Landscape Journal*, 29(2), pp. 107-123. doi: 10.3368/lj.29.2.107

Winqvist, C., Bengtsson, J., Öckinger, E., Aavik, T., Berendse, F., Clement, L.W., Fischer, C., Flohre, A., Geiger, F. and Liira, J. (2014) 'Species' traits influence ground beetle responses to farm and landscape level agricultural intensification in Europe', *Journal of Insect Conservation*, 18(5), pp. 837-846. doi: 10.1007/s10841-014-9690-0

Wise, D.H. and Molles Jr, M.C. (1979) 'Colonization of artificial substrates by stream insects: influence of substrate size and diversity', *Hydrobiologia*, 65(1), pp. 69-74. doi: 10.1007/BF00032721

Wolton, R.J., Chandler, P.J., Drake, C.M. and Stubbs, A.E. (2017) 'The relative importance of wet woodland and wet grassland for Diptera conservation: a case study from Devon, England', *Dipterist's Digest (Second Series),* 24(1), pp. 79-95.

Wong, M.H. (2003) 'Ecological restoration of mine degraded soils, with emphasis on metal contaminated soils', *Chemosphere*, 50(6), pp. 775-780. doi: 10.1016/S0045-6535(02)00232-1

Woodcock, B., Bal, S., Amy, S., Edwards, M., Redhead, J., Mountford, O., Gregory, S., Duffield, S., Macgregor, N. and Pywell, R. (2015) *Draft final report for the GREEN project (ECM6418): grassland restoration for ecological networks*. Available at: http://randd.defra.gov.uk/Document.aspx?Document=13458_LM0450_Final_report.pdf (Accessed: 9 October 2017).

Woodcock, B.A. (2005) 'Pitfall trapping in ecological studies', in Leather, S.R. (ed.) *Insect sampling in forest ecosystems*. Oxford: Blackwell Publishing, pp. 37-57.

Woodcock, B.A., Bullock, J.M., Shore, R.F., Heard, M.S., Pereira, M.G., Redhead, J., Ridding, L., Dean, H., Sleep, D., Henrys, P., Peyton, J., Hulmes, S., Hulmes, L., Sárospataki, M., Saure, C., Edwards, M., Genersch, E., Knäbe, S. and Pywell, R.F. (2017) 'Country-specific effects of neonicotinoid pesticides on honey bees and wild bees', *Science*, 356(6345), pp. 1393-1395. doi: 10.1126/science.aaa1190

Woodcock, B.A., Edwards, A.R., Lawson, C.S., Westbury, D.B., Brook, A.J., Harris, S.J., Brown, V.K. and Mortimer, S.R. (2008) 'Contrasting success in the restoration of plant and phytophagous beetle assemblages of species-rich mesotrophic grasslands', *Oecologia*, 154(4), pp. 773-783. doi: 10.1007/s00442-007-0872-2

Woodcock, B.A., Harrower, C., Redhead, J., Edwards, M., Vanbergen, A.J., Heard, M.S., Roy, D.B. and Pywell, R.F. (2014) 'National patterns of functional diversity and redundancy in predatory ground beetles and bees associated with key UK arable crops', *Journal of Applied Ecology*, 51(1), pp. 142-151. doi: 10.1111/1365-2664.12171

Woodcock, B.A., Potts, S.G., Tscheulin, T., Pilgrim, E., Ramsey, A.J., Harrison-Cripps, J., Brown, V.K. and Tallowin, J.R. (2009) 'Responses of invertebrate trophic level, feeding guild and body size to the management of improved grassland field margins', *Journal of Applied Ecology*, 46(4), pp. 920-929. doi: 10.1111/j.1365-2664.2009.01675.x

Woodcock, B.A., Potts, S.G., Westbury, D.B., Ramsay, A.J., Lambert, M., Harris, S.J. and Brown, V.K. (2007) 'The importance of sward architectural complexity in structuring predatory and phytophagous invertebrate assemblages', *Ecological Entomology*, 32(3), pp. 302-311. doi: 10.1111/j.1365-2311.2007.00869.x

Woodcock, B.A. and Pywell, R.F. (2009) 'Effects of vegetation structure and floristic diversity on detritivore, herbivore and predatory invertebrates within calcareous grasslands', *Biodiversity and Conservation*, 19(1), pp. 81-95. doi: 10.1007/s10531-009-9703-6

Woodcock, B.A., Redhead, J., Vanbergen, A.J., Hulmes, L., Hulmes, S., Peyton, J., Nowakowski, M., Pywell, R.F. and Heard, M.S. (2010) 'Impact of habitat type and landscape structure on biomass, species richness and functional diversity of ground beetles', *Agriculture, Ecosystems & Environment*, 139(1), pp. 181-186. doi: 10.1016/j.agee.2010.07.018

Woodcock, B.A., Watt, A.D. and Leather, S.R. (2003) 'Influence of management type on Diptera communities of coniferous plantations and deciduous woodlands', *Agriculture, Ecosystems & Environment*, 95(2), pp. 443-452. doi: 10.1016/S0167-8809(02)00220-7

Woods, R. (2012) 'Brownfield sites and moth diversity in the Tees Estuary', *Entomologist's Record and Journal of Variation,* 124(2), pp. 89-100.

Work, T.T., Buddle, C.M., Korinus, L.M. and Spence, J.R. (2002) 'Pitfall trap size and capture of three taxa of litter-dwelling arthropods: implications for biodiversity studies', *Environmental Entomology*, 31(3), pp. 438-448. doi: 10.1603/0046-225X-31.3.438

Wuellner, C.T. (1999) 'Nest site preference and success in a gregarious, ground-nesting bee *Dieunomia triangulifera*', *Ecological Entomology*, 24(4), pp. 471-479. doi: 10.1046/j.1365-2311.1999.00215.x

Yeo, P.F. and Corbet, S.A. (1995) *Solitary wasps.* Slough: The Richmond Publishing Co. Ltd. Naturalists' Handbooks 3.

Yu, D.W., Ji, Y., Emerson, B.C., Wang, X., Ye, C., Yang, C. and Ding, Z. (2012) 'Biodiversity soup: metabarcoding of arthropods for rapid biodiversity assessment and biomonitoring', *Methods in Ecology and Evolution*, 3(4), pp. 613-623. doi: 10.1111/j.2041-210X.2012.00198.x

Zulka, K.P., Milasowszky, N. and Lethmayer, C. (1997) 'Spider biodiversity potential of an ungrazed and a grazed inland salt meadow in the National Park 'Neusiedler See-Seewinkel' (Austria): implications for management (Arachnida: Araneae)', *Biodiversity and Conservation*, 6(1), pp. 75-88. doi: 10.1023/A:1018375615960

Zurbuchen, A., Landert, L., Klaiber, J., Müller, A., Hein, S. and Dorn, S. (2010) 'Maximum foraging ranges in solitary bees: only few individuals have the capability to cover long foraging distances', *Biological Conservation*, 143(3), pp. 669-676. doi: 10.1016/j.biocon.2009.12.003

Zuur, A.F., Ieno, E.N., Walker, N.J., Saveliev, A.A. and Smith, G.M. (2009) *Mixed effects models and extensions in ecology with R.* New York: Springer.

Chapter 9 – Appendices

Appendix 9.1

Table A9.1.1 Abundance of carabids at Canvey Wick, June-August 2015. Data based on captures in six pitfall traps, placed in areas of different management for nine days in each of June, July, and August 2015. Treatments were unmanaged controls (n=6), phase one cleared to mineral substrate in May 2014 (n=3), phase two cleared in May 2015 (n=6 plots), and open mosaic habitat reference areas (n=5). Ref. refers to codes for carabid species on the canonical correspondence analysis in Figure 2.7. Species were grouped as stenotopic indicators of early successional habitat; generalists associated with wetlands, woodland, or grassland; or eurytopic.

Ref.	Taxon	Authority	Group	Control	Phase One	Phase Two	Reference	Total
1	*Acupalpus brunnipes*	(Sturm)	Stenotopic	0	2	0	0	2
2	*Acupalpus dubius*	Schilsky	Generalist	8	8	30	1	47
3	*Acupalpus parvulus*	(Sturm)	Generalist	0	34	2	0	36
4	*Agonum emarginatum*	(Gyllenhal)	Generalist	2	1	0	0	3
5	*Agonum marginatum*	(Linnaeus)	Stenotopic	0	2	2	0	4
6	*Agonum muelleri*	(Herbst)	Eurytopic	0	4	0	0	4
7	*Amara aenea*	(De Geer)	Stenotopic	7	7	2	14	30
8	*Amara communis*	(Panzer)	Eurytopic	1	0	0	1	2
9	*Amara convexior*	Stephens	Stenotopic	8	31	13	4	56
10	*Amara curta*	Dejean	Stenotopic	0	0	0	1	1
11	*Amara eurynota*	(Panzer)	Stenotopic	4	1	0	3	8
12	*Amara familiaris*	(Duftschmid)	Stenotopic	0	3	0	9	12
13	*Amara lunicollis*	Schiödte	Eurytopic	9	0	0	3	12
14	*Amara montivaga*	Sturm	Stenotopic	0	0	0	1	1
15	*Amara ovata*	(Fabricius)	Eurytopic	1	0	0	3	4
16	*Amara similata*	(Gyllenhal)	Eurytopic	0	2	0	0	2
17	*Amara tibialis*	(Paykull)	Stenotopic	0	30	1	173	204
18	*Anchomenus dorsalis*	(Pontoppidan)	Eurytopic	0	1	0	0	1
19	*Anisodactylus binotatus*	(Fabricius)	Eurytopic	0	59	65	0	124

Ref.	Taxon	Authority	Group	Control	Phase One	Phase Two	Reference	Total
20	*Asaphidion flavipes*	(Linnaeus)	Stenotopic	0	13	0	0	13
21	*Badister bullatus*	(Schrank)	Eurytopic	0	0	0	1	1
22	*Bembidion assimile*	Gyllenhal	Generalist	0	1	1	0	2
23	*Bembidion biguttatum*	(Fabricius)	Generalist	1	0	0	0	1
24	*Bembidion femoratum*	Sturm	Stenotopic	0	0	16	0	16
25	*Bembidion illigeri*	Netolitzky	Stenotopic	0	0	2	0	2
26	*Bembidion lunulatum*	(Geoffroy)	Stenotopic	1	5	6	0	12
27	*Bembidion minimum*	(Fabricius)	Generalist	0	0	35	0	35
28	*Bembidion obtusum*	Audinet-Serville	Eurytopic	0	0	0	2	2
29	*Bembidion properans*	(Stephens)	Eurytopic	0	330	131	17	478
30	*Bembidion quadrimaculatum*	(Linnaeus)	Eurytopic	0	93	1167	1	1261
31	*Brachinus crepitans*	(Linnaeus)	Stenotopic	1	0	1	0	2
32	*Bradycellus verbasci*	(Duftschmid)	Eurytopic	0	1	0	0	1
33	*Calathus cinctus*	Motschulsky	Stenotopic	1	0	0	0	1
34	*Calathus fuscipes*	(Goeze)	Eurytopic	4	1	2	52	59
35	*Calathus rotundicollis*	Dejean	Generalist	3	0	0	0	3
36	*Carabus nemoralis*	Müller	Eurytopic	44	3	4	0	51
37	*Carabus violaceus*	Linnaeus	Eurytopic	16	4	0	3	23
38	*Chlaenius nigricornis*	(Fabricius)	Stenotopic	0	0	1	0	1
39	*Clivina fossor*	(Linnaeus)	Eurytopic	3	0	15	0	18
40	*Curtonotus aulicus*	(Panzer)	Eurytopic	1	0	6	6	13
41	*Curtonotus convexiusculus*	(Marsham)	Stenotopic	0	0	1	0	1
42	*Dyschirius aeneus*	(Dejean)	Stenotopic	0	34	9	0	43
43	*Dyschirius luedersi*	Wagner	Stenotopic	0	0	1	0	1
44	*Dyschirius politus*	(Dejean)	Stenotopic	0	46	6	0	52
45	*Elaphropus parvulus*	(Dejean)	Stenotopic	0	0	2	0	2
46	*Elaphrus riparius*	(Linnaeus)	Stenotopic	0	56	0	0	56
47	*Harpalus affinis*	(Schrank)	Eurytopic	4	14	15	35	68

Ref.	Taxon	Authority	Group	Control	Phase One	Phase Two	Reference	Total
48	*Harpalus anxius*	(Duftschmid)	Stenotopic	0	0	0	1	1
49	*Harpalus attenuatus*	Stephens	Stenotopic	0	0	0	3	3
50	*Harpalus rubripes*	(Duftschmid)	Stenotopic	0	0	0	10	10
51	*Harpalus rufipes*	(De Geer)	Eurytopic	0	3	72	1	76
52	*Harpalus tardus*	(Panzer)	Stenotopic	1	0	0	1	2
53	*Leistus ferrugineus*	(Linnaeus)	Eurytopic	16	0	0	0	16
54	*Loricera pilicornis*	(Fabricius)	Generalist	0	4	4	0	8
55	*Microlestes maurus*	(Sturm)	Eurytopic	0	3	0	6	9
56	*Microlestes minutulus*	(Goeze)	Stenotopic	0	2	0	2	4
57	*Nebria brevicollis*	(Fabricius)	Eurytopic	29	24	9	0	62
58	*Notiophilus biguttatus*	(Fabricius)	Generalist	44	2	3	0	49
59	*Notiophilus palustris*	(Duftschmid)	Generalist	46	1	1	0	48
60	*Notiophilus substriatus*	Waterhouse	Stenotopic	5	3	0	0	8
61	*Ophonus azureus*	(Fabricius)	Stenotopic	0	0	0	1	1
62	*Ophonus puncticeps*	Stephens	Eurytopic	0	0	0	6	6
63	*Ophonus rufibarbis*	(Fabricius)	Eurytopic	2	0	0	6	8
64	*Panagaeus bipustulatus*	(Fabricius)	Stenotopic	1	0	0	2	3
65	*Paradromius linearis*	(Olivier)	Stenotopic	0	0	1	1	2
66	*Poecilus cupreus*	(Linnaeus)	Stenotopic	0	6	6	0	12
67	*Pterostichus gracilis*	(Dejean)	Generalist	0	0	3	1	4
68	*Pterostichus macer*	(Marsham)	Generalist	0	0	1	0	1
69	*Pterostichus madidus*	(Fabricius)	Generalist	0	0	1	2	3
70	*Pterostichus melanarius*	(Illiger)	Eurytopic	79	4	47	2	132
71	*Pterostichus niger*	(Schaller)	Generalist	3	3	0	1	7
72	*Pterostichus nigrita*	(Paykull)	Generalist	0	2	49	0	51
73	*Pterostichus strenuus*	(Panzer)	Generalist	139	0	1	0	140
74	*Pterostichus vernalis*	(Panzer)	Eurytopic	0	0	1	0	1
75	*Stenolophus mixtus*	(Herbst)	Generalist	0	19	3	0	22

Ref.	Taxon	Authority	Group	Control	Phase One	Phase Two	Reference	Total
76	*Stenolophus teutonus*	(Schrank)	Stenotopic	0	64	2	0	66
77	*Stomis pumicatus*	(Panzer)	Generalist	33	4	6	1	44
78	*Syntomus foveatus*	(Geoffroy in Fourcroy)	Stenotopic	1	63	3	462	529
79	*Syntomus truncatellus*	(Linnaeus)	Stenotopic	3	2	0	0	5
80	*Trechus quadristriatus*	(Schrank)	Eurytopic	2	0	0	0	2

Table A9.1.2 Abundance of staphylinids at Canvey Wick, June-August 2015. Data based on captures in six pitfall traps, placed in areas of different management for nine days in each of June, July, and August 2015. Treatments were unmanaged controls (n=6), phase one cleared to mineral substrate in May 2014 (n=3), phase two cleared in May 2015 (n=6 plots), and open mosaic habitat reference areas (n=5). Ref. refers to codes for staphylinid species on the canonical correspondence analysis in Figure 2.8. Species were grouped as stenotopic indicators of early successional habitat; generalists associated with wetlands, woodland, or grassland; or eurytopic.

Ref.	Taxon	Authority	Group	Control	Phase One	Phase Two	Reference	Total
1	*Aleochara bipustulata*	(Linnaeus)	Eurytopic	0	4	103	2	109
2	*Aleochara brevipennis*	Gravenhorst	Eurytopic	0	78	95	0	173
3	*Aleochara curtula*	(Goeze)	Eurytopic	1	0	0	0	1
4	*Aleochara sparsa*	Heer	Eurytopic	3	0	0	0	3
5	*Aloconota gregaria*	(Erichson)	Eurytopic	3	1	31	0	35
6	*Amarochara forticornis*	(Lacordaire)	Generalist	2	1	0	0	3
7	*Amischa analis*	(Gravenhorst)	Eurytopic	6	5	5	7	23
8	*Amischa decipiens*	(Sharp)	Eurytopic	0	2	6	0	8
9	*Anotylus nitidulus*	(Gravenhorst)	Eurytopic	1	0	21	0	22
10	*Anotylus rugosus*	(Fabricius)	Eurytopic	6	5	272	1	284
11	*Anotylus sculpturatus*	(Gravenhorst)	Eurytopic	1	0	0	0	1
12	*Anotylus tetracarinatus*	(Block)	Eurytopic	1	0	0	0	1
13	*Atheta aquatica*	(Thomson)	Eurytopic	15	0	0	0	15
14	*Atheta castanoptera*	(Mannerheim)	Generalist	2	0	0	0	2
15	*Atheta vaga*	(Heer)	Eurytopic	1	0	0	0	1
16	*Autalia rivularis*	(Gravenhorst)	Eurytopic	3	0	0	0	3
17	*Bledius gallicus*	(Gravenhorst)	Stenotopic	0	63	5	0	68
18	*Brachygluta fossulata*	(Reichenbach)	Generalist	6	0	0	0	6
19	*Brundinia marina*	(Mulsant)	Unknown	0	0	2	0	2
20	*Callicerus rigidicornis*	(Erichson)	Generalist	25	2	5	0	32
21	*Calodera protensa*	Mannerheim	Generalist	10	5	9	2	26
22	*Carpelimus bilineatus*	Stephens	Stenotopic	0	3	14	0	17
23	*Carpelimus corticinus*	(Gravenhorst)	Stenotopic	0	2	9	0	11

Ref.	Taxon	Authority	Group	Control	Phase One	Phase Two	Reference	Total
24	*Carpelimus gracilis*	(Mannerheim)	Stenotopic	0	0	9	0	9
25	*Carpelimus incongruus*	Steel	Stenotopic	0	14	0	0	14
26	*Chaetida longicornis*	(Gravenhorst)	Eurytopic	1	0	0	0	1
27	*Cordalia obscura*	(Gravenhorst)	Generalist	1	0	0	0	1
28	*Cypha longicornis*	(Paykull)	Eurytopic	0	5	0	0	5
29	*Dinaraea angustula*	(Gyllenhal)	Eurytopic	4	5	11	1	21
30	*Drusilla canaliculata*	(Fabricius)	Eurytopic	239	4	2	11	256
31	*Gabrius appendiculatus*	Sharp	Generalist	0	1	0	0	1
32	*Gabrius breviventer*	(Sperk)	Generalist	0	10	18	1	29
33	*Gabrius nigritulus*	(Gravenhorst)	Stenotopic	0	0	1	0	1
34	*Gyrohypnus angustatus*	Stephens	Eurytopic	0	0	1	0	1
35	*Gyrohypnus wagneri*	(Scheerpeltz)	Unknown	3	0	0	0	3
36	*Gyrophaena bihamata*	Thomson	Generalist	1	0	0	0	1
37	*Gyrophaena fasciata*	(Marsham)	Generalist	1	0	0	0	1
38	*Ilyobates nigricollis*	(Paykull)	Unknown	0	1	0	0	1
39	*Ilyobates propinquus*	(Aubé)	Generalist	2	0	0	0	2
40	*Ischnosoma splendidum*	(Gravenhorst)	Generalist	2	0	0	0	2
41	*Lathrobium brunnipes*	(Fabricius)	Generalist	2	0	0	0	2
42	*Lathrobium geminum*	Kraatz	Eurytopic	35	2	15	1	53
43	*Liogluta longiuscula*	(Gravenhorst)	Eurytopic	2	0	0	0	2
44	*Liogluta pagana*	(Erichson)	Generalist	2	0	0	0	2
45	*Meotica exilis*	(Knoch)	Generalist	1	1	1	0	3
46	*Microdota indubia*	(Sharp)	Unknown	9	0	0	0	9
47	*Nehemitropia lividipennis*	(Mannerheim)	Eurytopic	0	0	8	0	8
48	*Neobisnius procerulus*	(Gravenhorst)	Stenotopic	0	0	29	0	29
49	*Neobisnius villosulus*	(Stephens)	Stenotopic	0	1	0	0	1
50	*Ocypus fuscatus*	(Gravenhorst)	Unknown	1	0	0	0	1
51	*Ocypus olens*	(Müller)	Eurytopic	2	1	1	1	5

Ref.	Taxon	Authority	Group	Control	Phase One	Phase Two	Reference	Total
52	*Oligota pumilio*	Kiesenwetter	Eurytopic	1	2	0	0	3
53	*Omalium septentrionis*	Thomson	Unknown	1	0	0	0	1
54	*Othius punctulatus*	(Goeze)	Generalist	2	0	7	0	9
55	*Othius subuliformis*	Stephens	Eurytopic	0	0	3	0	3
56	*Oxypoda brachyptera*	(Stephens)	Stenotopic	0	0	0	1	1
57	*Oxypoda brevicornis*	(Stephens)	Generalist	3	0	1	0	4
58	*Paederus littoralis*	Gravenhorst	Stenotopic	0	13	0	1	14
59	*Pella limbata*	(Paykull)	Generalist	40	1	1	14	56
60	*Philhygra elongatula*	(Gravenhorst)	Eurytopic	9	4	19	0	32
61	*Philhygra palustris*	(Kiesenwetter)	Stenotopic	0	0	8	0	8
62	*Philonthus cognatus*	Stephens	Eurytopic	0	6	1	0	7
63	*Philonthus fumarius*	(Gravenhorst)	Generalist	0	0	1	0	1
64	*Philonthus micans*	(Gravenhorst)	Generalist	0	0	1	0	1
65	*Philonthus succicola*	Thomson	Generalist	51	1	0	0	52
66	*Planeustomus palpalis*	(Erichson)	Stenotopic	0	1	0	0	1
67	*Platydracus stercorarius*	(Olivier)	Eurytopic	4	1	0	24	29
68	*Platystethus alutaceus*	Thomson	Stenotopic	0	1	0	0	1
69	*Platystethus nitens*	(Sahlberg)	Stenotopic	0	1	7	0	8
70	*Pselaphus heisei*	Herbst	Generalist	0	1	0	0	1
71	*Quedius boops*	(Gravenhorst)	Eurytopic	0	1	0	0	1
72	*Quedius curtipennis*	Bernhauer	Generalist	24	1	0	0	25
73	*Quedius levicollis*	(Brullé)	Generalist	0	2	0	0	2
74	*Quedius picipes*	(Mannerheim)	Eurytopic	11	1	0	1	13
75	*Rugilus erichsonii*	(Fauvel)	Generalist	0	0	1	0	1
76	*Rugilus orbiculatus*	(Paykull)	Stenotopic	0	0	3	0	3
77	*Rugilus rufipes*	Germar	Generalist	7	0	0	0	7
78	*Staphylinus dimidiaticornis*	Gemminger	Eurytopic	0	5	0	0	5
79	*Stenichnus bicolor*	(Denny)	Unknown	1	0	0	0	1

Ref.	Taxon	Authority	Group	Control	Phase One	Phase Two	Reference	Total
80	*Stenichnus scutellaris*	(Müller)	Unknown	12	0	1	0	13
81	*Stenus bimaculatus*	Gyllenhal	Generalist	1	0	0	0	1
82	*Stenus brunnipes*	Stephens	Eurytopic	0	1	0	0	1
83	*Stenus canaliculatus*	Gyllenhal	Stenotopic	0	10	0	0	10
84	*Stenus clavicornis*	(Scopoli)	Eurytopic	19	1	0	2	22
85	*Stenus juno*	(Paykull)	Generalist	0	1	0	0	1
86	*Stenus nanus*	Stephens	Eurytopic	0	0	0	5	5
87	*Stenus pusillus*	Stephens	Eurytopic	4	3	0	1	8
88	*Tachinus rufipes*	(Linnaeus)	Generalist	43	0	3	0	46
89	*Tachyporus dispar*	(Paykull)	Eurytopic	0	2	0	0	2
90	*Tachyporus hypnorum*	(Fabricius)	Eurytopic	0	14	10	0	24
91	*Tachyporus nitidulus*	(Fabricius)	Eurytopic	60	4	9	3	76
92	*Tachyporus pusillus*	Gravenhorst	Generalist	0	0	0	8	8
93	*Tachyporus scitulus*	Erichson	Generalist	0	0	0	1	1
94	*Tasgius ater*	(Gravenhorst)	Stenotopic	0	0	0	1	1
95	*Tasgius morsitans*	(Rossi)	Generalist	16	1	0	1	18
96	*Thecturota marchii*	(Dodero)	Stenotopic	0	0	4	0	4
97	*Xantholinus elegans*	(Olivier)	Stenotopic	10	0	6	3	19
98	*Xantholinus linearis*	(Olivier)	Eurytopic	13	4	8	4	29
99	*Xantholinus longiventris*	Heer	Eurytopic	2	13	21	0	36

Table A9.1.3 Abundance of spiders at Canvey Wick, June-August 2015. Data based on captures in six pitfall traps, placed in areas of different management for nine days in each of June, July, and August 2015. Treatments were unmanaged controls (n=6), phase one cleared to mineral substrate in May 2014 (n=3), phase two cleared in May 2015 (n=6 plots), and open mosaic habitat reference areas (n=5). Ref. refers to numerical codes for spider species on the canonical correspondence analysis in Figure 2.9. Species were grouped as stenotopic indicators of early successional habitat; generalists associated with wetlands, woodland, or grassland; or eurytopic.

Ref.	Taxon	Authority	Group	Control	Phase One	Phase Two	Reference	Total
1	*Agraecina striata*	(Kulczynski)	Generalist	89	8	20	4	121
2	*Agroeca inopina*	O.P.-Cambridge	Generalist	1	0	0	0	1
3	*Alopecosa pulverulenta*	(Clerck)	Eurytopic	9	0	2	30	41
4	*Antistea elegans*	(Blackwall)	Generalist	0	0	1	0	1
5	*Arctosa leopardus*	(Sundevall)	Stenotopic	1	38	33	1	73
6	*Argenna subnigra*	(O.P.-Cambridge)	Stenotopic	0	0	0	4	4
7	*Bathyphantes gracilis*	(Blackwall)	Eurytopic	10	5	1	0	16
8	*Bathyphantes parvulus*	(Westring)	Eurytopic	24	0	0	1	25
9	*Cheiracanthium virescens*	(Sundevall)	Stenotopic	0	2	1	2	5
10	*Clubiona comta*	C.L.Koch	Generalist	2	0	0	0	2
11	*Clubiona pallidula*	(Clerck)	Generalist	1	0	0	0	1
12	*Clubiona reclusa*	O.P.-Cambridge	Eurytopic	1	0	0	0	1
13	*Diplocephalus picinus*	(Blackwall)	Generalist	25	0	1	0	26
14	*Diplostyla concolor*	(Wider)	Eurytopic	12	0	1	0	13
15	*Dismodicus bifrons*	(Blackwall)	Eurytopic	1	0	0	0	1
16	*Drassodes cupreus*	(Blackwall)	Stenotopic	0	0	0	2	2
17	*Drassyllus pusillus*	(C.L.Koch)	Stenotopic	2	3	1	29	35
18	*Dysdera crocata*	C.L.Koch	Stenotopic	0	0	0	1	1
19	*Erigone arctica*	(White)	Stenotopic	0	0	2	0	2
20	*Erigone atra*	Blackwall	Eurytopic	0	41	56	0	97
21	*Erigone dentipalpis*	(Wider)	Eurytopic	3	124	145	1	273
22	*Euophrys frontalis*	(Walckenaer)	Eurytopic	0	0	1	0	1
23	*Hahnia nava*	(Blackwall)	Eurytopic	6	0	0	4	10

Ref.	Taxon	Authority	Group	Control	Phase One	Phase Two	Reference	Total
24	*Haplodrassus signifer*	(C.L.Koch)	Stenotopic	0	1	0	8	9
25	*Heliophanus flavipes*	(Hahn)	Generalist	0	0	0	3	3
26	*Linyphia hortensis*	Sundevall	Generalist	1	0	0	0	1
27	*Maso sundevalli*	(Westring)	Generalist	13	0	0	0	13
28	*Meioneta beata*	(O.P.-Cambridge)	Generalist	1	3	0	0	4
29	*Meioneta rurestris*	(C.L.Koch)	Eurytopic	0	0	1	0	1
30	*Meioneta saxatilis*	(Blackwall)	Eurytopic	2	0	0	0	2
31	*Micaria pulicaria*	(Sundevall)	Eurytopic	1	17	11	4	33
32	*Micrargus subaequalis*	(Westring)	Generalist	0	1	3	0	4
33	*Microneta viaria*	(Blackwall)	Generalist	10	0	0	0	10
34	*Oedothorax apicatus*	(Blackwall)	Eurytopic	1	184	572	0	757
35	*Oedothorax fuscus*	(Blackwall)	Eurytopic	0	29	9	0	38
36	*Oedothorax retusus*	(Westring)	Eurytopic	0	58	18	0	76
37	*Ostearius melanopygius*	(O.P.-Cambridge)	Eurytopic	0	0	4	1	5
38	*Ozyptila brevipes*	(Hahn)	Generalist	1	0	0	0	1
39	*Ozyptila praticola*	(C.L.Koch)	Generalist	11	0	0	0	11
40	*Ozyptila sanctuaria*	(O.P.-Cambridge)	Generalist	2	4	0	7	13
41	*Ozyptila simplex*	(O.P.-Cambridge)	Stenotopic	33	50	1	5	89
42	*Pachygnatha degeeri*	Sundevall	Eurytopic	8	20	1	0	29
43	*Pardosa agrestis*	(Westring)	Stenotopic	1	112	142	1	256
44	*Pardosa nigriceps*	(Thorell)	Eurytopic	0	0	0	3	3
45	*Pardosa palustris*	(Linnaeus)	Eurytopic	0	3	0	3	6
46	*Pardosa prativaga*	(L.Koch)	Eurytopic	0	7	3	1	11
47	*Pardosa pullata*	(Clerck)	Eurytopic	4	7	0	23	34
48	*Pardosa saltans*	Töpfer-Hofmann	Generalist	1	0	0	0	1
49	*Pelecopsis parallela*	(Wider)	Generalist	0	0	0	14	14
50	*Phrurolithus festivus*	(C.L.Koch)	Eurytopic	29	22	26	13	90
51	*Pirata piraticus*	(Clerck)	Generalist	0	0	3	0	3

Ref.	Taxon	Authority	Group	Control	Phase One	Phase Two	Reference	Total
52	*Pocadicnemis juncea*	Locket	Eurytopic	5	1	1	0	7
53	*Porrhomma microphthalmum*	(O.P.-Cambridge)	Eurytopic	0	0	3	0	3
54	*Talavera aequipes*	(O.P.-Cambridge)	Stenotopic	1	1	1	0	3
55	*Tenuiphantes tenuis*	(Blackwall)	Eurytopic	44	1	11	5	61
56	*Tiso vagans*	(Blackwall)	Eurytopic	8	0	0	0	8
57	*Trachyzelotes pedestris*	(C.L.Koch)	Stenotopic	0	1	1	15	17
58	*Trochosa ruricola*	(De Geer)	Eurytopic	0	5	11	0	16
59	*Trochosa terricola*	Thorell	Eurytopic	45	2	5	13	65
60	*Walckenaeria acuminata*	Blackwall	Eurytopic	0	0	1	0	1
61	*Walckenaeria antica*	(Wider)	Eurytopic	3	0	0	2	5
62	*Walckenaeria atrotibialis*	(O.P.-Cambridge)	Generalist	27	0	0	0	27
63	*Walckenaeria vigilax*	(Blackwall)	Generalist	0	6	8	0	14
64	*Xysticus cristatus*	(Clerck)	Eurytopic	0	1	0	3	4
65	*Xysticus kochi*	Thorell	Stenotopic	0	4	1	2	7
66	*Zelotes apricorum*	(L.Koch)	Stenotopic	8	0	4	5	17
67	*Zelotes latreillei*	(Simon)	Stenotopic	0	0	0	6	6
68	*Zodarion italicum*	(Canestrini)	Stenotopic	0	1	1	41	43
69	*Zora spinimana*	(Sundevall)	Generalist	29	0	1	1	31

Appendix 9.2

9.2.1 Introduction

Brownfield biodiversity research has focused on plants, beetles, spiders, bugs and aculeate Hymenoptera. Information about Diptera is lacking, which reflects wider neglect in applied ecology, particularly community studies except for Syrphidae (Orford *et al.*, 2015). This may result from greater difficulty in species-level identification for some Diptera families, despite the potential for ecological interpretation based on a wide variety of trophic strategies (Woodcock *et al.*, 2003). There has also been a lack of recognition for the role of Diptera in ecosystem service provision, such as non-syrphids in pollination studies (Ssymank *et al.*, 2008). A review of unpublished consultancy surveys suggested limited conservation interest for UK brownfield Diptera (Gibson, 1998). Although data could be affected by variation in surveyor activity (Isaac *et al.*, 2014), it was argued that Syrphidae, with less than 3% of conservation priority species recorded, were well-studied and representative of all Diptera (Gibson, 1998).

Unpublished records from Canvey Wick indicate that some previously developed land could have high value for Diptera conservation. 65 of 605 Diptera species reported between 2000 and 2017 had provisional or full conservation status, or 10.7% of the total recorded fauna. This provides a potentially rich pool of species to identify significant responses of Dipteran communities to conservation management. The aim in this study was to assess the conservation value at Canvey Wick of grassland, and the grassland/scrub interface. The focus was on calypterate Diptera, as these represent a major component of species richness, abundance, and functional diversity for flies of grassland and scrub (Peng *et al.*, 1992). This was intended to create added value for the research in relation to potential future management, and to improve understanding of Diptera at Open Mosaic Habitat in relation to habitat structure.

9.2.2 Methods

Sampling for this study was completed at Canvey Wick on 9-10 May 2017. Six spatially interspersed replicates of each habitat were sampled between 0930 and 1300, with the survey order randomised. This was intended to reduce the

potential effect of diel flight patterns (Peng *et al.*, 1992). A sample comprised 100 sweeps of a 40 cm diameter, 75 cm deep white cotton organdy net, attached to a 10 foot pole. Adult Diptera in the calypterate families were removed from the net with a pooter and killed with ethyl acetate, before identification in the laboratory. This survey protocol was repeated on both dates. Calypterate Diptera were determined to species in the families Anthomyiidae (Ackland, 2012), Calliphoridae (Rognes, 1991), Fanniidae (Rozkošný *et al.*, 1997), Muscidae (Gregor *et al.*, 2016), Rhinophoridae (Van Emden, 1954), Sarcophagidae (Pape, 1987) and Tachinidae (Belshaw, 1993). Only male Anthomyiidae were identified, due to the difficulty of classifying female specimens from morphological characteristics. Provisional conservation status for the species recorded were taken from the most recently available reviews (Falk and Pont, 1996; Falk and Pont, 2017).

9.2.3 Results

217 individuals were recorded in the families Anthomyiidae, Calliphoridae, Fanniidae, Muscidae, Rhinophoridae, Sarcophagidae, and Tachinidae (Table A9.2.1). There were 66 individuals in the grassland samples, and 151 individuals in the scrub samples. Seven species were unique to the grassland samples, 29 species were unique to the scrub samples, and 8 species were captured in both habitats (Figure A9.2.1). 26 species were only recorded as single individuals, and those present in low numbers could have been missed in grassland or scrub. The importance of the two habitats might also change through the year, particularly with availability of floral resources. Despite these limitations, the results suggest that the scrub/grassland interface might be an important part of the habitat mosaic for calypterate Diptera. This is consistent with other research. Emergent hedgerow trees were important in structuring distribution and abundance of Diptera, likely offering shelter, food, as a swarming marker, and during mate searching (Peng *et al.*, 1992). There were also distinct communities of Dolichopodidae in dunes, associated with open dry habitats, scrubby vegetation, and dune woodland (Pollet and Grootaert, 1996).

Table A9.2.1 Capture data for calypterate Diptera from grassland (G, n=6) and scrub/grassland interface (I, n=6) at Canvey Wick on 9-10 May 2017. Samples were collected from 100 sweeps of a 40 cm diameter, 75 cm deep white cotton organdy net, attached to a 10 foot pole. This was repeated in each monitoring plot, between 0930 and 1300, on both dates.

Family	Taxon	Authority	G	I	Total
Anthomyiidae	*Anthomyia confusanea*	Michelsen	1	1	1
Anthomyiidae	*Anthomyia pluvialis*	(Linnaeus)	0	1	1
Anthomyiidae	*Anthomyia procellaris*	Rondani	0	3	3
Anthomyiidae	*Botanophila fugax*	(Zetterstedt)	0	1	1
Anthomyiidae	*Delia platura*	(Meigen)	6	1	7
Anthomyiidae	*Hylemya variata*	(Fallén)	0	1	1
Anthomyiidae	*Pegoplata annulata*	(Pandellé)	0	1	1
Anthomyiidae	*Subhylemyia longula*	(Fallén)	1	0	1
Calliphoridae	*Calliphora vicina*	Robineau-Desvoidy	0	1	1
Calliphoridae	*Pollenia griseotomentosa*	(Jacentkowský)	0	1	1
Fanniidae	*Fannia armata*	(Meigen)	0	1	1
Fanniidae	*Fannia fuscula*	(Fallén)	1	1	2
Fanniidae	*Fannia genualis*	(Stein)	0	1	1
Fanniidae	*Fannia lustrator*	(Harris)	0	8	8
Fanniidae	*Fannia rondanii*	(Strobl)	0	3	3
Fanniidae	*Fannia serena*	(Fallén)	0	1	1
Fanniidae	*Fannia sociella*	(Zetterstedt	0	1	1
Fanniidae	*Fannia sociella*	(Zetterstedt)	0	34	34
Muscidae	*Azelia zetterstedtii*	Rondani	0	4	4
Muscidae	*Coenosia infantula*	Rondani	23	5	28
Muscidae	*Coenosia pumila*	(Fallén)	2	0	2
Muscidae	*Coenosia testacea*	(Robineau-Desvoidy)	19	7	26
Muscidae	*Hebecnema fumosa*	(Meigen)	0	1	1
Muscidae	*Hebecnema nigra*	(Robineau-Desvoidy)	0	1	1
Muscidae	*Hebecnema nigricolor*	(Fallén)	0	9	9
Muscidae	*Hebecnema umbratica*	(Meigen)	0	3	3
Muscidae	*Helina impuncta*	(Fallén)	1	3	4
Muscidae	*Helina lasiophthalma*	(Macquart)	1	12	13
Muscidae	*Helina reversio*	(Harris)	3	0	3
Muscidae	*Hydrotaea armipes*	(Fallén)	0	1	1
Muscidae	*Hydrotaea floccosa*	Macquart	0	1	1
Muscidae	*Morellia simplex*	(Loew)	0	7	7
Muscidae	*Musca autumnalis*	De Geer	0	18	18
Muscidae	*Muscina levida*	(Harris)	0	1	1
Muscidae	*Muscina prolapsa*	(Harris)	0	1	1
Muscidae	*Mydaea ancilla*	(Meigen)	0	1	1
Muscidae	*Neomyia cornicina*	(Fabricius)	1	0	1
Muscidae	*Phaonia fuscata*	(Fallén)	0	3	3
Muscidae	*Phaonia subventa*	(Harris)	0	2	2
Rhinophoridae	*Phyto melanocephala*	(Meigen)	1	5	6
Sarcophagidae	*Sarcophaga haemorrhoa*	Meigen	0	1	1
Tachinidae	*Gymnosoma nitens*	Meigen	2	0	2
Tachinidae	*Lydella stabulans*	(Meigen)	0	1	1
Tachinidae	*Lydina aenea*	(Meigen)	1	0	1

Family	Taxon	Authority	G	I	Total
Tachinidae	*Macquartia dispar*	(Fallén)	1	2	3
Tachinidae	*Phania funesta*	(Meigen)	1	0	1
Tachinidae	*Subclytia rotundiventris*	(Fallén)	0	1	1
Tachinidae	*Zophomyia temula*	(Scopoli)	1	0	1

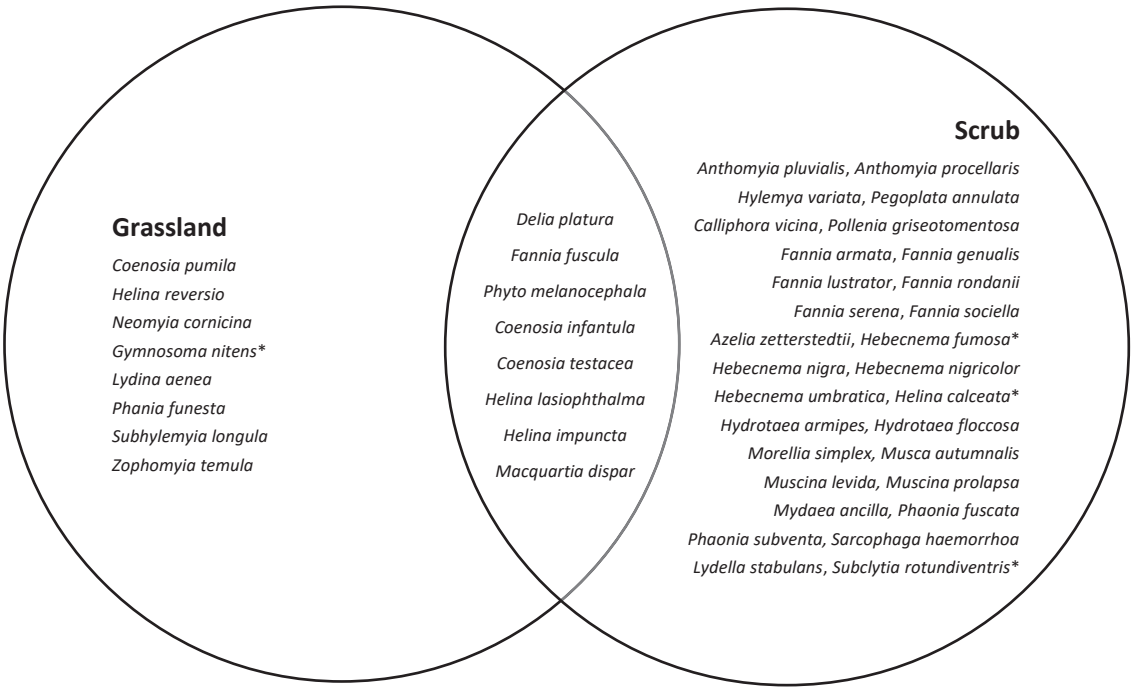

Figure A9.2.1 Overlap between species of calypterate Diptera from sweep net samples in grassland and scrub at Canvey Wick on 9-10 May 2017. Asterisk (*) by species name indicates provisional UK conservation priority status (Falk and Pont, 1996; Falk and Pont, 2017).

9.2.4 Discussion

For species with provisional UK conservation status at Canvey Wick, *Hebecnema fumosa* (Muscidae, provisionally Nationally Scarce) is part of a genus with predatory larvae, mostly occurring in the dung of ungulates (Skidmore, 1985). *Helina calceata* (Muscidae, provisionally Nationally Scarce) is poorly known, though some localities are dry grassland with scrub, or on sandy soils (Falk and Pont, 2017). *Subclytia rotundiventris* (Tachinidae, provisionally Near Threatened) is a parasite of shieldbugs (Hemiptera, Acanthosomatidae), including species associated with *Betula* (Falk and Pont, 1996). Conservation for this species was considered to require trees and shrubs as part of a mosaic of vegetation types at occupied sites, including wet heaths and grasslands (Falk and Pont, 1996). *Gymnosoma nitens* (Tachinidae, provisionally Near Threatened) is a parasite of the shieldbug *Sciocoris cursitans*, which is most frequent at warm, dry sites on chalk or sand in southern England (Falk and Pont, 1996). The fly has consistently been associated with unshaded, well-drained soils and open vegetation structure, although foodplants of the shieldbug host are unknown (Harvey, 2001).

The ecology of conservation priority Diptera in this study emphasises the importance of a mosaic of brownfield grassland and scrub, to accommodate diverse life-history requirements. Each individual habitat is expected to support specialist species, while others may depend on these in combination. Some Muscidae inhabit accumulated organic matter during larval development, whereas adults feed on nectar or pollen (Skidmore, 1985). This is similar to the concept of partial habitats, developed to explain contrasting foraging and nesting requirements of aculeate Hymenoptera (Westrich, 1996). A study of Diptera in wet grassland and woodland also suggested that these should be managed as complementary rather than conflicting habitats (Wolton *et al.*, 2017). The difficulty is to translate this general advice into management prescriptions. Activity ranges of Diptera are poorly studied, so the importance of relative proximity in habitats is unclear. The shape, edge length, and orientation of scrub blocks are also likely to be important. Seasonal abundance of *Episyrphus balteatus* (Syrphidae) was particularly linked to aspect, and varied depending on temperature and availability of floral resources (Sarthou *et al.*, 2005).

Appendix 9.3

Table A9.3.1 Carabid functional trait data, extracted from literature. Size = median length (mm) from the range given for males and females (Luff, 2007). SPR: 1 = main period of species activity in spring (Luff, 2007). AUT: 1 = main period of species activity in autumn (Luff, 2007). PRE: 1 = predatory adult feeding guild (Ribera *et al.*, 2001; Harvey *et al.*, 2008). OMN: 1 = omnivorous adult feeding guild (Ribera *et al.*, 2001; Harvey *et al.*, 2008). PHY: phytophagous adult feeding guild, including frequently graminivorous species (Ribera *et al.*, 2001; Harvey *et al.*, 2008; Vanbergen *et al.*, 2010). BRA: 1 = brachypterous or apterous wing development (Luff, 2007). MAC: 1 = macropterous wing development (Luff, 2007). DIM: 1 = wing-dimorphic species (Luff, 2007). CON: UK conservation status based on the most recent published review (Telfer, 2016).

Tribe	Taxon	Authority	SIZE	SPR	AUT	PRE	OMN	PHY	BRA	DIM	MAC	CON
Bembidiini	*Asaphidion flavipes*	(Linnaeus)	4.30	1	0	1	0	0	0	0	1	1
Bembidiini	*Bembidion assimile*	Gyllenhal	3.25	1	0	1	0	0	0	1	0	0
Bembidiini	*Bembidion biguttatum*	(Fabricius)	3.25	1	0	1	0	0	0	0	1	0
Bembidiini	*Bembidion femoratum*	Sturm	4.65	1	0	1	0	0	0	0	1	0
Bembidiini	*Bembidion illigeri*	Netolitzky	4.65	1	0	1	0	0	0	0	1	0
Bembidiini	*Bembidion lunulatum*	(Geoffroy in Fourcroy)	3.75	1	0	1	0	0	0	0	1	0
Bembidiini	*Bembidion minimum*	(Fabricius)	2.85	1	0	1	0	0	0	0	1	0
Bembidiini	*Bembidion obtusum*	Audinet-Serville	3.15	1	0	1	0	0	0	1	0	0
Bembidiini	*Bembidion properans*	(Stephens)	3.95	1	0	1	0	0	0	1	0	0
Bembidiini	*Bembidion quadrimaculatum*	(Linnaeus)	3.10	1	0	1	0	0	0	0	1	0
Bembidiini	*Elaphropus parvulus*	(Dejean)	2.00	1	0	1	0	0	0	0	1	1
Brachinini	*Brachinus crepitans*	(Linnaeus)	7.75	1	0	0	1	0	0	0	1	1
Carabini	*Carabus nemoralis*	Müller, O.F.	23.00	0	1	1	0	0	1	0	0	0
Carabini	*Carabus violaceus*	Linnaeus	25.00	0	1	1	0	0	1	0	0	0
Chlaeniini	*Chlaenius nigricornis*	(Fabricius)	11.00	1	0	1	0	0	0	0	1	0
Elaphrini	*Elaphrus riparius*	(Linnaeus)	7.00	1	0	1	0	0	0	0	1	0
Harpalini	*Acupalpus brunnipes*	(Sturm)	3.25	1	0	0	1	0	0	0	1	1
Harpalini	*Acupalpus dubius*	Schilsky	2.65	1	0	0	1	0	0	0	1	0

Tribe	Taxon	Authority	SIZE	SPR	AUT	PRE	OMN	PHY	BRA	DIM	MAC	CON
Harpalini	*Acupalpus parvulus*	(Sturm)	3.50	1	0	0	1	0	0	0	1	0
Harpalini	*Anisodactylus binotatus*	(Fabricius)	11.50	1	0	0	1	0	0	0	1	0
Harpalini	*Bradycellus verbasci*	(Duftschmid)	4.50	0	1	0	0	1	0	1	0	0
Harpalini	*Harpalus affinis*	(Schrank)	10.50	1	0	0	0	1	0	0	1	0
Harpalini	*Harpalus anxius*	(Duftschmid)	7.25	1	0	0	0	1	0	0	1	1
Harpalini	*Harpalus attenuatus*	Stephens	8.00	1	0	0	0	1	0	0	1	1
Harpalini	*Harpalus rubripes*	(Duftschmid)	10.25	1	0	0	0	1	0	0	1	0
Harpalini	*Harpalus rufipes*	(De Geer)	13.50	0	1	0	0	1	0	0	1	0
Harpalini	*Harpalus tardus*	(Panzer)	9.75	1	0	0	0	1	0	0	1	0
Harpalini	*Ophonus azureus*	(Fabricius)	5.00	1	0	0	0	1	0	1	0	0
Harpalini	*Ophonus puncticeps*	Stephens	7.00	0	1	0	0	1	0	0	1	0
Harpalini	*Ophonus rufibarbis*	(Fabricius)	8.00	1	0	0	0	1	0	0	1	0
Harpalini	*Stenolophus mixtus*	(Herbst)	5.50	1	0	0	1	0	0	0	1	0
Harpalini	*Stenolophus teutonus*	(Schrank)	6.00	1	0	0	1	0	0	0	1	0
Lebiini	*Microlestes maurus*	(Sturm)	2.55	1	0	1	0	0	1	0	0	0
Lebiini	*Microlestes minutulus*	(Goeze)	3.25	1	0	1	0	0	0	0	1	0
Lebiini	*Paradromius linearis*	(Olivier)	5.00	1	0	1	0	0	0	1	0	0
Lebiini	*Syntomus foveatus*	(Geoffroy in Fourcroy)	3.50	1	0	1	0	0	1	0	0	0
Lebiini	*Syntomus truncatellus*	(Linnaeus)	3.05	1	0	1	0	0	0	1	0	1
Licinini	*Badister bullatus*	(Schrank)	5.55	1	0	1	0	0	0	0	1	0
Loricerini	*Loricera pilicornis*	(Fabricius)	7.00	1	0	1	0	0	0	0	1	0
Nebriini	*Leistus ferrugineus*	(Linnaeus)	7.00	0	1	1	0	0	1	0	0	0
Nebriini	*Nebria brevicollis*	(Fabricius)	12.50	0	1	1	0	0	0	0	1	0
Notiophilini	*Notiophilus biguttatus*	(Fabricius)	5.50	1	0	1	0	0	0	1	0	0
Notiophilini	*Notiophilus palustris*	(Duftschmid)	5.00	1	0	1	0	0	0	0	1	0
Notiophilini	*Notiophilus substriatus*	Waterhouse	5.00	1	0	1	0	0	0	0	1	0
Panagaeini	*Panagaeus bipustulatus*	(Fabricius)	7.00	1	0	0	1	0	0	0	1	1
Platynini	*Agonum emarginatum*	(Gyllenhal)	8.25	1	0	1	0	0	0	0	1	0

Tribe	Taxon	Authority	SIZE	SPR	AUT	PRE	OMN	PHY	BRA	DIM	MAC	CON
Platynini	*Agonum marginatum*	(Linnaeus)	9.60	1	0	1	0	0	0	0	1	0
Platynini	*Agonum muelleri*	(Herbst)	8.00	1	0	1	0	0	0	0	1	0
Platynini	*Anchomenus dorsalis*	(Pontoppidan)	7.00	1	0	1	0	0	0	0	1	0
Pterostichini	*Poecilus cupreus*	(Linnaeus)	12.00	1	0	1	0	0	0	0	1	0
Pterostichini	*Pterostichus gracilis*	(Dejean)	9.00	1	0	0	1	0	0	0	1	1
Pterostichini	*Pterostichus macer*	(Marsham)	12.25	1	0	0	1	0	0	0	1	0
Pterostichini	*Pterostichus madidus*	(Fabricius)	16.00	0	1	0	1	0	1	0	0	1
Pterostichini	*Pterostichus melanarius*	(Illiger)	15.00	0	1	0	1	0	0	1	0	0
Pterostichini	*Pterostichus niger*	(Schaller)	18.50	0	1	0	1	0	0	0	1	0
Pterostichini	*Pterostichus nigrita*	(Panzer)	10.50	1	0	0	1	0	0	0	1	0
Pterostichini	*Pterostichus strenuus*	(Panzer)	6.60	1	0	0	1	0	0	1	0	0
Pterostichini	*Pterostichus vernalis*	(Panzer)	6.75	1	0	0	1	0	0	1	0	0
Pterostichini	*Stomis pumicatus*	(Panzer)	7.50	1	0	1	0	0	1	0	0	0
Scaritini	*Clivina fossor*	(Linnaeus)	6.40	1	0	1	0	0	0	1	0	0
Scaritini	*Dyschirius aeneus*	(Dejean)	3.35	1	0	1	0	0	0	0	1	0
Scaritini	*Dyschirius luedersi*	Wagner	3.60	1	0	1	0	0	0	0	1	0
Scaritini	*Dyschirius politus*	(Dejean)	4.25	1	0	1	0	0	0	0	1	1
Sphodrini	*Calathus cinctus*	Motschulsky	7.25	0	1	0	1	0	0	1	0	0
Sphodrini	*Calathus fuscipes*	(Goeze)	12.00	0	1	0	1	0	1	0	0	0
Sphodrini	*Calathus rotundicollis*	Dejean	9.50	1	0	0	1	0	0	1	0	0
Trechini	*Trechus quadristriatus*	(Schrank)	3.85	0	1	1	0	0	0	0	1	0
Zabrini	*Amara aenea*	(De Geer)	7.65	1	0	0	0	1	0	0	1	0
Zabrini	*Amara communis*	(Panzer)	7.00	1	0	0	0	1	0	0	1	0
Zabrini	*Amara convexior*	Stephens	7.35	1	0	0	0	1	0	0	1	0
Zabrini	*Amara curta*	Dejean	6.35	1	0	0	0	1	0	0	1	1
Zabrini	*Amara eurynota*	(Panzer)	11.05	1	0	0	0	1	0	0	1	0
Zabrini	*Amara familiaris*	(Duftschmid)	6.40	1	0	0	0	1	0	0	1	0
Zabrini	*Amara lunicollis*	Schiödte	8.45	1	0	0	0	1	0	0	1	0

Tribe	Taxon	Authority	SIZE	SPR	AUT	PRE	OMN	PHY	BRA	DIM	MAC	CON
Zabrini	*Amara montivaga*	Sturm	8.45	1	0	0	0	1	0	0	1	1
Zabrini	*Amara ovata*	(Fabricius)	8.75	1	0	0	0	1	0	0	1	0
Zabrini	*Amara similata*	(Gyllenhal)	8.75	1	0	0	0	1	0	0	1	0
Zabrini	*Amara tibialis*	(Panzer)	5.20	1	0	0	0	1	0	0	1	0
Zabrini	*Curtonotus aulicus*	(Panzer)	12.50	0	1	0	0	1	0	0	1	0
Zabrini	*Curtonotus convexiusculus*	(Marsham)	11.90	0	1	0	0	1	0	0	1	0

Table A9.3.2 Staphylinid functional trait data, extracted from literature. Size = median length (mm) from the range given for males and females (Lott, 2009; Lott and Anderson, 2011; Freude *et al.*, 1974). LIT: 1 = adult resource preference for litter (Lott, 2009; Lott and Anderson, 2011; Freude *et al.*, 1974; García-Tejero and Taboada, 2016). MYC: 1 = mycetophilous adult resource preference (Lott, 2009; Lott and Anderson, 2011; Freude *et al.*, 1974; García-Tejero and Taboada, 2016). MYR: 1 = myrmecophilous adult resource preference (Lott, 2009; Lott and Anderson, 2011; Freude *et al.*, 1974; García-Tejero and Taboada, 2016). SAP: 1 = adult resource preference for dung or carrion (Lott, 2009; Lott and Anderson, 2011; Freude *et al.*, 1974; García-Tejero and Taboada, 2016). CON: UK conservation status based on the most recent available review (Hyman and Parsons, 1994).

Tribe	Taxon	Authority	Size	LIT	MYC	MYR	PHY	SAP	CON
Aleocharini	*Aleochara bipustulata*	(Linnaeus, 1761)	3.25	0	0	0	0	1	0
Aleocharini	*Aleochara brevipennis*	Gravenhorst, 1806	4.50	1	0	0	0	0	0
Aleocharini	*Aleochara curtula*	(Goeze, 1777)	6.00	0	0	0	0	1	0
Aleocharini	*Aleochara sparsa*	Heer, 1839	3.25	1	0	0	0	0	0
Athetini	*Aloconota gregaria*	(Erichson, 1839)	3.50	1	0	0	0	0	0
Athetini	*Amischa analis*	(Gravenhorst, 1802)	2.10	1	0	0	0	0	0
Athetini	*Amischa decipiens*	(Sharp, 1869)	2.10	1	0	0	0	0	0
Athetini	*Atheta aquatica*	(Thomson, C.G., 1852)	3.80	1	0	0	0	0	0
Athetini	*Atheta castanoptera*	(Mannerheim, 1830)	3.90	0	1	0	0	0	0
Athetini	*Atheta vaga*	(Heer, 1839)	2.65	0	1	0	0	0	0
Athetini	*Brundinia marina*	(Mulsant & Rey, 1853)	2.70	1	0	0	0	0	0
Athetini	*Callicerus rigidicornis*	(Erichson, 1839)	4.25	1	0	0	0	0	0
Athetini	*Chaetida longicornis*	(Gravenhorst, 1802)	3.25	1	0	0	0	0	0
Athetini	*Dinaraea angustula*	(Gyllenhal, 1810)	3.35	1	0	0	0	0	0
Athetini	*Liogluta longiuscula*	(Gravenhorst, 1802)	4.20	1	0	0	0	0	0
Athetini	*Liogluta pagana*	(Erichson, 1839)	4.75	1	0	0	0	0	1
Athetini	*Microdota indubia*	(Sharp, 1869)	2.10	1	0	0	0	0	0
Athetini	*Nehemitropia lividipennis*	(Mannerheim, 1830)	3.25	1	0	0	0	0	0
Athetini	*Philhygra palustris*	(Kiesenwetter, 1844)	2.45	1	0	0	0	0	0
Autaliini	*Autalia rivularis*	(Gravenhorst, 1802)	1.90	1	0	0	0	0	0
Brachyglutini	*Brachygluta fossulata*	(Reichenbach, 1816)	1.80	1	0	0	0	0	0
Cyrtoscydmini	*Stenichnus bicolor*	(Denny, 1825)	1.60	1	0	0	0	0	0

Tribe	Taxon	Authority	Size	LIT	MYC	MYR	PHY	SAP	CON
Cyrtoscydmini	*Stenichnus scutellaris*	(Müller, P.W.J. & Kunze, 1822)	1.60	1	0	0	0	0	0
Falagriini	*Cordalia obscura*	(Gravenhorst, 1802)	2.40	1	0	0	0	0	0
Homalotini	*Gyrophaena bihamata*	Thomson, C.G., 1867	1.90	0	1	0	0	0	0
Homalotini	*Thecturota marchii*	(Dodero, 1922)	1.60	1	0	0	0	0	0
Hygronomini	*Oligota pumilio*	Kiesenwetter, 1858	1.15	1	0	0	0	0	0
Hypocyphtini	*Cypha longicornis*	(Paykull, 1800)	1.20	1	0	0	0	0	0
Lomechusini	*Drusilla canaliculata*	(Fabricius, 1787)	4.40	1	0	0	0	0	0
Lomechusini	*Pella limbata*	(Paykull, 1789)	5.00	0	0	1	0	0	0
Lomechusini	*Philhygra elongatula*	(Gravenhorst, 1802)	3.25	1	0	0	0	0	0
Mycetoporini	*Ischnosoma splendidum*	(Gravenhorst, 1806)	4.25	0	1	0	0	0	0
Omaliini	*Omalium septentrionis*	Thomson, C.G., 1856	3.25	1	0	0	0	0	0
Othiini	*Othius punctulatus*	(Goeze, 1777)	11.00	1	0	0	0	0	0
Othiini	*Othius subuliformis*	Stephens, 1833	5.00	1	0	0	0	0	0
Oxypodini	*Amarochara forticornis*	(Lacordaire, 1835)	2.25	1	0	0	0	0	0
Oxypodini	*Calodera protensa*	Mannerheim, 1830	3.10	1	0	0	0	0	1
Oxypodini	*Ilyobates nigricollis*	(Paykull, 1800)	4.75	1	0	0	0	0	1
Oxypodini	*Ilyobates propinquus*	(Aubé, 1850)	5.75	1	0	0	0	0	1
Oxypodini	*Meotica exilis*	(Knoch in Gravenhorst, 1806)	1.55	1	0	0	0	0	0
Oxypodini	*Oxypoda brachyptera*	(Stephens, 1832)	2.10	1	0	0	0	0	0
Oxypodini	*Oxypoda brevicornis*	(Stephens, 1832)	2.85	1	0	0	0	0	0
Oxytelini	*Anotylus nitidulus*	(Gravenhorst, 1802)	2.65	0	0	0	0	1	0
Oxytelini	*Anotylus rugosus*	(Fabricius, 1775)	4.75	1	0	0	0	0	0
Oxytelini	*Anotylus sculpturatus*	(Gravenhorst, 1806)	4.00	0	0	0	0	1	0
Oxytelini	*Anotylus tetracarinatus*	(Block, 1799)	2.15	0	0	0	0	1	0
Oxytelini	*Platystethus alutaceus*	Thomson, C.G., 1861	4.05	1	0	0	0	0	0
Oxytelini	*Platystethus nitens*	(Sahlberg, C.R., 1832)	2.75	1	0	0	0	0	0
Paederini	*Lathrobium brunnipes*	(Fabricius, 1792)	8.75	1	0	0	0	0	0
Paederini	*Lathrobium geminum*	Kraatz, 1857	9.00	1	0	0	0	0	0

Tribe	Taxon	Authority	Size	LIT	MYC	MYR	PHY	SAP	CON
Paederini	*Paederus littoralis*	Gravenhorst, 1802	8.50	1	0	0	0	0	0
Paederini	*Rugilus erichsonii*	(Fauvel, 1867)	4.25	1	0	0	0	0	0
Paederini	*Rugilus orbiculatus*	(Paykull, 1789)	4.50	1	0	0	0	0	0
Paederini	*Rugilus rufipes*	Germar, 1836	6.00	1	0	0	0	0	0
Pselaphini	*Pselaphus heisei*	Herbst, 1792	2.68	1	0	0	0	0	0
Staphylinini	*Gabrius appendiculatus*	Sharp, 1910	4.50	1	0	0	0	0	0
Staphylinini	*Gabrius breviventer*	(Sperk, 1835)	4.25	1	0	0	0	0	0
Staphylinini	*Gabrius nigritulus*	(Gravenhorst, 1802)	4.75	1	0	0	0	0	0
Staphylinini	*Neobisnius procerulus*	(Gravenhorst, 1806)	5.00	1	0	0	0	0	1
Staphylinini	*Neobisnius villosulus*	(Stephens, 1833)	5.00	1	0	0	0	0	0
Staphylinini	*Ocypus fuscatus*	(Gravenhorst, 1802)	15.00	1	0	0	0	0	1
Staphylinini	*Ocypus olens*	(Müller, O.F., 1764)	27.50	1	0	0	0	0	0
Staphylinini	*Philonthus cognatus*	Stephens, 1832	9.50	1	0	0	0	0	0
Staphylinini	*Philonthus fumarius*	(Gravenhorst, 1806)	8.75	1	0	0	0	0	1
Staphylinini	*Philonthus micans*	(Gravenhorst, 1802)	6.75	1	0	0	0	0	0
Staphylinini	*Philonthus succicola*	Thomson, C.G., 1860	11.50	1	0	0	0	0	0
Staphylinini	*Platydracus stercorarius*	(Olivier, 1795)	14.50	1	0	0	0	0	0
Staphylinini	*Quedius boops*	(Gravenhorst, 1802)	5.00	1	0	0	0	0	0
Staphylinini	*Quedius curtipennis*	Bernhauer, 1908	12.00	1	0	0	0	0	0
Staphylinini	*Quedius levicollis*	(Brullé, 1832)	12.00	1	0	0	0	0	0
Staphylinini	*Quedius picipes*	(Mannerheim, 1830)	10.00	1	0	0	0	0	0
Staphylinini	*Staphylinus dimidiaticornis*	Gemminger, 1851	19.50	1	0	0	0	0	0
Staphylinini	*Tasgius ater*	(Gravenhorst, 1802)	16.50	1	0	0	0	0	0
Staphylinini	*Tasgius morsitans*	(Rossi, 1790)	14.00	1	0	0	0	0	0
Steninae	*Stenus bimaculatus*	Gyllenhal, 1810	6.75	1	0	0	0	0	0
Steninae	*Stenus brunnipes*	Stephens, 1833	3.50	1	0	0	0	0	0
Steninae	*Stenus canaliculatus*	Gyllenhal, 1827	4.00	1	0	0	0	0	0
Steninae	*Stenus clavicornis*	(Scopoli, 1763)	5.75	1	0	0	0	0	0

Tribe	Taxon	Authority	Size	LIT	MYC	MYR	PHY	SAP	CON
Steninae	*Stenus juno*	(Paykull, 1789)	6.50	1	0	0	0	0	0
Steninae	*Stenus nanus*	Stephens, 1833	2.38	1	0	0	0	0	0
Steninae	*Stenus pusillus*	Stephens, 1833	2.50	1	0	0	0	0	0
Tachyporini	*Tachinus rufipes*	(Linnaeus, 1758)	5.75	1	0	0	0	0	0
Tachyporini	*Tachyporus dispar*	(Paykull, 1789)	3.75	1	0	0	0	0	0
Tachyporini	*Tachyporus hypnorum*	(Fabricius, 1775)	3.50	1	0	0	0	0	0
Tachyporini	*Tachyporus nitidulus*	(Fabricius, 1781)	2.50	1	0	0	0	0	0
Tachyporini	*Tachyporus pusillus*	Gravenhorst, 1806	2.50	1	0	0	0	0	0
Tachyporini	*Tachyporus scitulus*	Erichson, 1839	2.50	1	0	0	0	0	1
Thinobiini	*Bledius gallicus*	(Gravenhorst, 1806)	4.50	0	0	0	1	0	0
Thinobiini	*Carpelimus bilineatus*	Stephens, 1834	3.50	0	0	0	1	0	0
Thinobiini	*Carpelimus corticinus*	(Gravenhorst, 1806)	2.25	0	0	0	1	0	0
Thinobiini	*Carpelimus gracilis*	(Mannerheim, 1830)	1.90	1	0	0	1	0	0
Thinobiini	*Carpelimus incongruus*	Steel, 1969	2.45	1	0	0	1	0	0
Thinobiini	*Planeustomus palpalis*	(Erichson, 1839)	2.25	1	0	0	0	0	0
Xantholinini	*Gyrohypnus angustatus*	Stephens, 1833	6.50	1	0	0	0	0	0
Xantholinini	*Gyrohypnus wagneri*	(Scheerpeltz, 1926)	6.50	1	0	0	0	0	0
Xantholinini	*Gyrophaena fasciata*	(Marsham, 1802)	2.10	0	1	0	0	0	0
Xantholinini	*Xantholinus elegans*	(Olivier, 1795)	10.50	1	0	0	0	0	0
Xantholinini	*Xantholinus linearis*	(Olivier, 1795)	7.50	1	0	0	0	0	0
Xantholinini	*Xantholinus longiventris*	Heer, 1839	7.75	1	0	0	0	0	0

Table A9.3.3 Spider functional trait data, extracted from literature. Size = median length (mm) from the range given for females (Roberts, 1985; Roberts, 1987). MOI: Standardised niche parameter between 0 for species in moistest habitats, and 1 for species in driest habitats (Entling *et al.*, 2007). SHA: Standardised niche parameter between 0 for species in the most shaded habitats, and 1 for species in the most open habitats (Entling *et al.*, 2007). BAL: 1 = species specific record of ballooning (Bell *et al.*, 2005; Blandenier, 2009). AMB: 1 = ambush hunting strategy (Uetz *et al.*, 1999). RUN: 1 = running hunting strategy (Uetz *et al.*, 1999). WEB: 1 = web hunting strategy (Uetz *et al.*, 1999). CON: 1 = UK conservation status based on the most recent published review (Merrett, 1990).

Family	Taxon	Authority	Size	MOI	SHA	BAL	AMB	RUN	WEB	CON
Clubionidae	*Cheiracanthium virescens*	(Sundevall)	7.00	0.55	0.11	0	0	0	1	0
Clubionidae	*Clubiona comta*	C.L.Koch	4.75	0.32	0.58	0	0	0	1	0
Clubionidae	*Clubiona pallidula*	(Clerck)	9.00	0.35	0.54	0	0	0	1	0
Clubionidae	*Clubiona reclusa*	O.P.-Cambridge	7.50	0.27	0.28	0	0	0	1	0
Dictynidae	*Argenna subnigra*	(O.P.-Cambridge)	2.13	0.51	0.15	1	0	0	1	0
Dysderidae	*Dysdera crocata*	C.L.Koch	13.00	0.40	0.33	0	0	1	0	0
Gnaphosidae	*Drassodes cupreus*	(Blackwall)	13.50	0.62	0.29	0	0	1	0	0
Gnaphosidae	*Drassyllus pusillus*	(C.L.Koch)	4.50	0.50	0.21	0	0	1	0	0
Gnaphosidae	*Haplodrassus signifer*	(C.L.Koch)	8.50	0.60	0.40	0	0	1	0	0
Gnaphosidae	*Micaria pulicaria*	(Sundevall)	3.60	0.36	0.22	0	0	1	0	0
Gnaphosidae	*Trachyzelotes pedestris*	(C.L.Koch)	7.50	0.60	0.34	1	0	1	0	1
Gnaphosidae	*Zelotes apricorum*	(L.Koch)	7.75	0.64	0.42	1	0	1	0	0
Gnaphosidae	*Zelotes latreillei*	(Simon)	7.50	0.50	0.26	0	0	1	0	0
Hahniidae	*Antistea elegans*	(Blackwall)	2.75	0.23	0.27	0	0	0	1	0
Hahniidae	*Hahnia nava*	(Blackwall)	1.75	0.56	0.23	0	0	0	1	0
Linyphiidae	*Bathyphantes gracilis*	(Blackwall)	1.95	0.28	0.31	1	0	0	1	0
Linyphiidae	*Bathyphantes parvulus*	(Westring)	2.25	0.35	0.38	1	0	0	1	0
Linyphiidae	*Diplocephalus picinus*	(Blackwall)	2.25	0.35	0.28	1	0	0	1	0
Linyphiidae	*Diplostyla concolor*	(Wider)	1.70	0.30	0.52	1	0	0	1	0
Linyphiidae	*Dismodicus bifrons*	(Blackwall)	2.60	0.32	0.41	1	0	0	1	0
Linyphiidae	*Erigone arctica*	(White)	3.10	0.07	0.06	1	0	0	1	0
Linyphiidae	*Erigone atra*	Blackwall	2.30	0.29	0.29	1	0	0	1	0

Family	Taxon	Authority	Size	MOI	SHA	BAL	AMB	RUN	WEB	CON
Linyphiidae	*Erigone dentipalpis*	(Wider)	2.20	0.31	0.23	1	0	0	1	0
Linyphiidae	*Linyphia hortensis*	Sundevall	4.50	0.36	0.62	1	0	0	1	0
Linyphiidae	*Maso sundevalli*	(Westring)	1.55	0.37	0.49	0	0	0	1	0
Linyphiidae	*Meioneta beata*	(O.P.-Cambridge)	1.85	0.51	0.34	0	0	0	1	0
Linyphiidae	*Meioneta rurestris*	(C.L.Koch)	2.00	0.38	0.25	1	0	0	1	0
Linyphiidae	*Meioneta saxatilis*	(Blackwall)	2.00	0.33	0.43	0	0	0	1	0
Linyphiidae	*Micrargus subaequalis*	(Westring)	4.75	0.38	0.20	1	0	0	1	0
Linyphiidae	*Microneta viaria*	(Blackwall)	2.75	0.36	0.58	1	0	0	1	0
Linyphiidae	*Oedothorax apicatus*	(Blackwall)	2.80	0.28	0.19	1	0	0	1	0
Linyphiidae	*Oedothorax fuscus*	(Blackwall)	2.65	0.22	0.20	1	0	0	1	0
Linyphiidae	*Oedothorax retusus*	(Westring)	2.60	0.18	0.25	1	0	0	1	0
Linyphiidae	*Ostearius melanopygius*	(O.P.-Cambridge)	2.30	0.31	0.17	1	0	0	1	0
Linyphiidae	*Pelecopsis parallela*	(Wider)	1.75	0.31	0.15	1	0	0	1	0
Linyphiidae	*Pocadicnemis juncea*	Locket & Millidge	1.95	0.29	0.26	0	0	0	1	0
Linyphiidae	*Porrhomma microphthalmum*	(O.P.-Cambridge)	1.95	0.29	0.24	1	0	0	1	0
Linyphiidae	*Tenuiphantes tenuis*	(Blackwall)	2.65	0.31	0.31	1	0	0	1	0
Linyphiidae	*Tiso vagans*	(Blackwall)	1.95	0.34	0.24	1	0	0	1	0
Linyphiidae	*Walckenaeria acuminata*	Blackwall	3.50	0.37	0.5	1	0	0	1	0
Linyphiidae	*Walckenaeria antica*	(Wider)	2.30	0.44	0.35	1	0	0	1	0
Linyphiidae	*Walckenaeria atrotibialis*	(Wider)	2.50	0.32	0.46	1	0	0	1	0
Linyphiidae	*Walckenaeria vigilax*	(Blackwall)	2.30	0.28	0.18	0	0	0	1	0
Liocranidae	*Agraecina striata*	(Kulczynski)	5.00	0.38	0.25	0	0	1	0	1
Liocranidae	*Phrurolithus festivus*	(C.L.Koch)	2.75	0.43	0.29	0	0	1	0	0
Lycosidae	*Alopecosa pulverulenta*	(Clerck)	8.25	0.42	0.29	1	0	1	0	0
Lycosidae	*Arctosa leopardus*	(Sundevall)	9.00	0.28	0.14	1	0	0	1	0
Lycosidae	*Pardosa agrestis*	(Westring)	7.50	0.30	0.12	1	0	1	0	1
Lycosidae	*Pardosa nigriceps*	(Thorell)	6.00	0.36	0.34	1	0	1	0	0
Lycosidae	*Pardosa palustris*	(Linnaeus)	5.25	0.32	0.20	1	0	1	0	0

Family	Taxon	Authority	Size	MOI	SHA	BAL	AMB	RUN	WEB	CON
Lycosidae	*Pardosa prativaga*	(L.Koch)	5.00	0.26	0.20	1	0	1	0	0
Lycosidae	*Pardosa pullata*	(Clerck)	5.00	0.36	0.26	1	0	1	0	0
Lycosidae	*Pardosa saltans*	Töpfer-Hofmann	5.50	0.42	0.49	0	0	1	0	0
Lycosidae	*Pirata piraticus*	(Clerck)	6.75	0.21	0.22	1	0	1	0	0
Lycosidae	*Trochosa ruricola*	(De Geer)	11.5	0.32	0.22	0	0	1	0	0
Lycosidae	*Trochosa terricola*	Thorell	10.5	0.43	0.45	1	0	1	0	0
Salticidae	*Euophrys frontalis*	(Walckenaer)	4.00	0.48	0.36	0	1	0	0	0
Salticidae	*Heliophanus flavipes*	(Hahn)	5.50	0.53	0.17	0	1	0	0	0
Salticidae	*Talavera aequipes*	(O.P.-Cambridge)	2.50	0.59	0.28	1	1	0	0	0
Tetragnathidae	*Pachygnatha degeeri*	Sundevall	3.38	0.38	0.27	1	0	0	1	0
Thomisidae	*Ozyptila brevipes*	(Hahn)	3.50	0.41	0.46	0	1	0	0	0
Thomisidae	*Ozyptila praticola*	(C.L.Koch)	3.50	0.34	0.44	0	1	0	0	0
Thomisidae	*Ozyptila sanctuaria*	(O.P.-Cambridge)	3.50	0.51	0.28	1	1	0	0	0
Thomisidae	*Ozyptila simplex*	(O.P.-Cambridge)	4.50	0.39	0.16	1	1	0	0	0
Thomisidae	*Xysticus cristatus*	(Clerck)	7.00	0.42	0.24	1	1	0	0	0
Thomisidae	*Xysticus kochi*	Thorell	7.00	0.42	0.2	1	1	0	0	0
Zoridae	*Zora spinimana*	(Sundevall)	5.75	0.39	0.43	1	0	1	0	0

Appendix 9.4

Table A9.4.1 Morphological variables for carabid and staphylinid species at Canvey Wick experimental ground disturbance plots in 2016, derived from measured characters (Tables 4.2 and 4.3). Abbreviations for morphological variables: LOG/PEL medial length of pronotum plus length of elytra, log-transformed(x+1); EYW/HEW diameter of the eye divided by the width of the head; ANL/PEL length of the antenna divided by LOG/PEL; MAL/PEL length of the longest mandible divided by LOG/PEL; PRL/PRW length of pronotum divided by width of pronotum; ELW/ELL maximum width of elytra divided by maximum length of elytra; TRL/HFL length of metatrochanter divided by length of metafemur; HFL/HLL length of metafemur divided by total length of hind leg; HTL/HLL length of metatibia divided by total length of hind leg; HRL/HLL length of metatarsi divided by total length of hind leg; FFL/FLL length of profemur divided by total length of front leg; FTL/FLL length of protibia divided by total length of front leg ; FRL/FLL length of protarsi divided by total length of front leg. WING wing development (b = apterous or brachypterous, d = dimorphic, m = macropterous).

Taxon	LOGPEL	EYW/HEW	ANL/PEL	MAL/PEL	PRL/PRW	ELW/ELL	TFL/HFL	HFL/HLL	HTL/HLL	HRL/HLL	FFL/FLL	FTL/FLL	FTL/FLL	WING
Carabidae														
Acupalpus dubius	1.6458	0.1277	0.1304	0.6619	0.7778	0.6628	0.5056	0.3735	0.3572	0.2694	0.3869	0.3460	0.2670	m
Acupalpus parvulus	1.7272	0.1708	0.1267	0.5955	0.8253	0.6620	0.5093	0.3526	0.3726	0.2749	0.3925	0.3419	0.2656	m
Agonum emarginatum	2.0556	0.2000	0.1104	0.7171	0.7854	0.7005	0.3957	0.3370	0.3399	0.3231	0.3638	0.3242	0.3121	m
Agonum marginatum	2.1503	0.2122	0.1113	0.6132	0.7327	0.6576	0.3658	0.3513	0.3529	0.2958	0.3774	0.3426	0.2800	m
Amara aenea	2.0265	0.1549	0.1055	0.4160	0.5906	0.7240	0.5556	0.3168	0.3652	0.3180	0.3803	0.3181	0.3016	m
Amara communis	1.9818	0.0948	0.1246	0.4897	0.6048	0.7301	0.5659	0.3210	0.3521	0.3269	0.3252	0.2853	0.3895	m
Amara convexior	2.0485	0.1250	0.1224	0.4547	0.5959	0.7280	0.5597	0.3388	0.3527	0.3085	0.3759	0.3334	0.2907	m
Amara lunicollis	2.0451	0.1066	0.1110	0.4477	0.6071	0.7568	0.5229	0.3207	0.3574	0.3219	0.3551	0.3343	0.3106	m
Anisodactylus binotatus	2.2156	0.1334	0.1313	0.4539	0.6608	0.6523	0.5310	0.3320	0.3602	0.3079	0.3689	0.3443	0.2868	m
Bembidion lunulatum	1.7563	0.2540	0.1103	0.6054	0.7566	0.6862	0.4459	0.3781	0.3404	0.2815	0.3823	0.3438	0.2739	m
Bembidion minimum	1.6294	0.2567	0.1111	0.5659	0.7471	0.6626	0.4727	0.3796	0.3250	0.2954	0.3981	0.3376	0.2643	m
Bembidion properans	1.7644	0.2930	0.1052	0.5851	0.7011	0.6920	0.4753	0.3589	0.3409	0.3003	0.3815	0.3479	0.2706	d
Bembidion quadrimaculatum	1.6201	0.2166	0.1219	0.7256	0.7275	0.7571	0.4054	0.3595	0.3386	0.3019	0.3801	0.3479	0.2721	m
Brachinus crepitans	1.9342	0.2106	0.1361	0.8480	0.9089	0.6938	0.3033	0.3524	0.3311	0.3164	0.3658	0.3361	0.2981	m
Bradycellus harpalinus	1.8143	0.2100	0.1285	0.6099	0.7376	0.6495	0.4754	0.3615	0.3734	0.2651	0.3644	0.3745	0.2611	d
Bradycellus verbasci	1.8267	0.2105	0.1387	0.4814	0.7486	0.6794	0.4390	0.3629	0.3744	0.2626	0.3761	0.3679	0.2560	m
Calathus fuscipes	2.2494	0.1841	0.1133	0.6420	0.8134	0.6355	0.4319	0.3146	0.3433	0.3421	0.3542	0.3360	0.3099	b

Taxon	LOGPEL	EYW/HEW	ANL/PEL	MAL/PEL	PRL/PRW	ELW/ELL	TFL/HFL	HFL/HLL	HTL/HLL	HRL/HLL	FFL/FLL	FTL/FLL	FTL/FLL	WING
Carabus nemoralis	2.4922	0.2011	0.1534	0.6430	0.6780	0.7231	0.3335	0.3147	0.3410	0.3443	0.3515	0.3190	0.3295	b
Carabus violaceus	2.5610	0.2129	0.1351	0.6485	0.7060	0.6294	0.2933	0.3234	0.3590	0.3176	0.3694	0.3086	0.3219	b
Clivina fossor	1.9119	0.1482	0.1302	0.4621	0.8524	0.5443	0.4772	0.3660	0.3437	0.2903	0.3548	0.4180	0.2272	d
Curtonotus aulicus	2.2378	0.1486	0.1457	0.4529	0.6155	0.7314	0.4688	0.3351	0.3335	0.3314	0.3665	0.3509	0.2826	m
Dyschirius aeneus	1.6530	0.2610	0.1161	0.3814	0.9249	0.6346	0.4167	0.3639	0.3620	0.2741	0.3711	0.4009	0.2280	m
Dyschirius politus	1.7607	0.2227	0.1151	0.3666	0.9703	0.5271	0.4147	0.3370	0.3723	0.2907	0.3587	0.4015	0.2398	m
Elaphrus riparius	2.0043	0.1529	0.1447	0.4421	0.8832	0.7530	0.4001	0.3743	0.3300	0.2957	0.3889	0.3441	0.2670	m
Harpalus affinis	2.1305	0.1513	0.1554	0.5009	0.6667	0.6840	0.6292	0.3391	0.3615	0.2994	0.3616	0.3457	0.2926	m
Harpalus rubripes	2.1584	0.0906	0.1874	0.4305	0.6364	0.7248	0.6298	0.3587	0.3816	0.2597	0.4039	0.3619	0.2342	m
Harpalus tardus	2.1469	0.1038	0.1695	0.4303	0.6267	0.7011	0.6592	0.3445	0.3592	0.2963	0.3649	0.3510	0.2840	m
Microlestes maurus	1.4739	0.1529	0.1451	0.7908	0.7599	0.7798	0.4097	0.3586	0.3149	0.3265	0.3714	0.3346	0.2939	m
Nebria brevicollis	2.2261	0.1885	0.1229	0.6759	0.6373	0.6638	0.2850	0.3345	0.3297	0.3358	0.3557	0.3524	0.2920	m
Notiophilus palustris	1.8831	0.2583	0.1332	0.4131	0.6203	0.6192	0.3395	0.3465	0.3283	0.3252	0.3836	0.3182	0.2982	b
Notiophilus substriatus	1.8854	0.2188	0.1285	0.3833	0.6638	0.5796	0.3285	0.3354	0.3354	0.3292	0.3624	0.3378	0.2998	m
Paradromius linearis	1.8196	0.1842	0.1199	0.6293	0.9003	0.5624	0.3704	0.3520	0.3390	0.3090	0.3826	0.3479	0.2695	d
Poecilus cupreus	2.2544	0.2330	0.1152	0.5665	0.7005	0.6470	0.5245	0.3217	0.3340	0.3443	0.3587	0.3136	0.3277	m
Pterostichus gracilis	2.0767	0.1843	0.1258	0.6105	0.7758	0.6775	0.5949	0.3300	0.3514	0.3186	0.3589	0.3411	0.3000	m
Pterostichus melanarius	2.3676	0.1342	0.1495	0.5207	0.7209	0.6445	0.4744	0.3429	0.3448	0.3123	0.3795	0.3330	0.2876	d
Pterostichus niger	2.4321	0.1716	0.1250	0.6342	0.7853	0.6174	0.4319	0.3532	0.3500	0.2968	0.3638	0.3391	0.2971	m
Pterostichus nigrita	2.1958	0.1821	0.1307	0.5625	0.7385	0.6004	0.5518	0.3390	0.3519	0.3091	0.3654	0.3460	0.2886	m
Pterostichus strenuus	1.9217	0.1781	0.1108	0.5747	0.7675	0.6757	0.4750	0.3525	0.3509	0.2966	0.3773	0.3510	0.2718	d
Stenolophus teutonus	1.9789	0.1474	0.1349	0.5216	0.8284	0.6493	0.5487	0.3616	0.3701	0.2683	0.4007	0.3386	0.2607	m
Stenolophus mixtus	1.9331	0.1842	0.1262	0.6117	0.7443	0.6004	0.5576	0.3508	0.3676	0.2816	0.3925	0.3528	0.2547	m
Syntomus foveatus	1.6399	0.2118	0.1270	0.6345	0.6913	0.6954	0.3977	0.3576	0.3272	0.3152	0.3770	0.3606	0.2623	m
Staphylinidae														
Aleochara bipustulata	1.3097	0.1043	0.1959	0.8391	0.7411	1.6583	0.4821	0.3581	0.3414	0.3005	0.4301	0.3518	0.2182	m
Aleochara brevipennis	1.4807	0.1125	0.1790	0.7278	0.6897	1.7780	0.4783	0.3676	0.3408	0.2916	0.4142	0.3592	0.2265	m
Aloconota gregaria	1.2511	0.0859	0.1844	1.1539	0.9121	1.3800	0.4165	0.3651	0.3651	0.2697	0.4157	0.3730	0.2114	m

Taxon	LOGPEL	EYW/HEW	ANL/PEL	MAL/PEL	PRL/PRW	ELW/ELL	TFL/HFL	HFL/HLL	HTL/HLL	HRL/HLL	FFL/FLL	FTL/FLL	FTL/FLL	WING
Amischa analis	1.0212	0.0686	0.1927	1.0000	0.8333	1.4571	0.3441	0.3981	0.3344	0.2675	0.4059	0.3762	0.2178	m
Amischa decipiens	1.0068	0.0690	0.1969	1.0341	0.8571	1.9000	0.3438	0.3951	0.3580	0.2469	0.4028	0.3732	0.2239	m
Anotylus nitidulus	1.1725	0.1039	0.2200	0.7200	0.6739	1.2727	0.3333	0.3939	0.4040	0.2020	0.4024	0.3780	0.2195	m
Anotylus rugosus	1.4008	0.1018	0.2615	0.9140	0.6818	1.4332	0.3003	0.4114	0.4114	0.1772	0.4096	0.3976	0.1928	m
Bledius gallicus	1.4583	0.1861	0.2311	0.6501	0.8621	1.0000	0.2587	0.3843	0.3984	0.2174	0.4097	0.3665	0.2238	m
Brachygluta fossulata	1.1959	0.1905	0.1683	1.1158	0.8333	1.5357	0.3077	0.3662	0.4085	0.2254	0.3810	0.3810	0.2381	m
Callicerus rigidicornis	1.3733	0.1117	0.2156	1.2777	0.8422	1.3918	0.3890	0.3545	0.3717	0.2737	0.4220	0.3687	0.2092	m
Calodera protensa	1.2224	0.0973	0.2380	1.1595	1.0100	1.2270	0.3636	0.3646	0.3757	0.2597	0.4047	0.3774	0.2179	m
Carpelimus corticinus	1.0892	0.1177	0.2461	0.8914	0.7574	1.2017	0.2750	0.4353	0.4013	0.1635	0.4389	0.3990	0.1620	m
Carpelimus incongruus	1.0864	0.0938	0.2750	0.7969	0.9048	1.2619	0.2593	0.4148	0.4097	0.1756	0.4138	0.3966	0.1897	m
Dinaraea angustula	1.2452	0.0706	0.1957	1.0841	0.8401	1.3516	0.4088	0.4067	0.3734	0.2200	0.4462	0.3523	0.2015	m
Drusilla canaliculata	1.3747	0.1008	0.2150	1.1636	1.1154	1.4350	0.4704	0.3434	0.3484	0.3083	0.4054	0.3611	0.2335	b
Gabrius breviventer	1.3558	0.1191	0.2410	0.8267	1.2793	1.1131	0.3404	0.3642	0.3463	0.2895	0.4510	0.2819	0.2671	m
Ischnosoma splendidum	1.4615	0.1367	0.1026	0.8152	0.8911	0.9148	0.3495	0.2821	0.2282	0.4897	0.3813	0.2554	0.3634	m
Lathrobium brunnipes	1.6255	0.1474	0.2391	0.8784	1.1786	1.0573	0.3022	0.3986	0.3464	0.2551	0.4206	0.3391	0.2403	b
Lathrobium geminum	1.5966	0.1216	0.2392	1.0057	1.0536	0.9993	0.3417	0.3962	0.3498	0.2540	0.4418	0.3318	0.2264	m
Lobrathium multipunctum	1.3747	0.0693	0.2301	1.0736	1.2535	1.1892	0.3158	0.3931	0.3612	0.2457	0.4146	0.3476	0.2378	d
Neobisnius procerulus	1.4242	0.1110	0.2447	0.8266	1.4128	1.0090	0.3313	0.3698	0.3385	0.2917	0.4369	0.2984	0.2647	m
Ocypus fuscatus	1.8277	0.1572	0.2773	0.8046	1.0146	1.3257	0.3581	0.3405	0.3458	0.3137	0.4363	0.2964	0.2673	m
Ocypus olens	2.0728	0.1249	0.3821	0.8662	0.9112	1.3565	0.3313	0.3500	0.3431	0.3070	0.4079	0.3100	0.2821	m
Oligota pumilio	0.8605	0.0679	0.1972	0.8234	0.5944	1.7376	0.4230	0.3598	0.4009	0.2393	0.4182	0.3931	0.1887	m
Paederus littoralis	1.6322	0.5481	0.2907	0.8226	0.9839	1.0714	0.2618	0.3399	0.3666	0.2935	0.3754	0.3632	0.2614	m
Pella limbata	1.4216	0.0633	0.2483	1.1760	0.7544	1.5963	0.3662	0.3612	0.3747	0.2642	0.4245	0.3528	0.2227	m
Philhygra elongatula	1.2352	0.1053	0.1972	1.0868	0.8632	0.7105	0.3821	0.3686	0.3584	0.2730	0.4630	0.3519	0.1852	m
Philonthus carbonarius	1.5183	0.0962	0.2435	0.9270	0.9829	1.6154	0.3333	0.3333	0.3333	0.3333	0.4643	0.1964	0.3393	m
Philonthus cognatus	1.7003	0.1148	0.2171	0.8594	0.9842	1.5712	0.3017	0.3415	0.3214	0.3371	0.3758	0.2985	0.3258	m
Platydracus stercorarius	1.8622	0.0941	0.3351	0.7072	0.8658	1.4876	0.3665	0.3645	0.3292	0.3064	0.4454	0.2864	0.2681	m
Platystethus alutaceus	1.3096	0.0882	0.3676	0.9076	0.7556	1.6471	0.3436	0.4112	0.3832	0.2056	0.4000	0.4000	0.2000	m

Taxon	LOGPEL	EYW/HEW	ANL/PEL	MAL/PEL	PRL/PRW	ELW/ELL	TFL/HFL	HFL/HLL	HTL/HLL	HRL/HLL	FFL/FLL	FTL/FLL	FTL/FLL	WING
Platystethus nitens	1.1649	0.0724	0.3210	1.0543	0.7370	1.2548	0.3219	0.4110	0.3808	0.2082	0.4020	0.3747	0.2234	m
Quedius curtipennis	1.7579	0.1833	0.2471	0.6792	0.9142	1.3512	0.3098	0.3610	0.3381	0.3009	0.4489	0.2693	0.2818	m
Quedius levicollis	1.8149	0.1629	0.2633	0.6631	0.8880	1.3184	0.3053	0.3597	0.3393	0.3010	0.4389	0.2690	0.2920	m
Quedius picipes	1.6828	0.1840	0.2861	0.8153	0.8815	1.4180	0.2737	0.3591	0.3530	0.2879	0.4140	0.3025	0.2835	m
Quedius semiobscurus	1.6283	0.2474	0.1960	0.6913	0.9679	1.4501	0.3306	0.3523	0.3318	0.3159	0.4248	0.2795	0.2957	m
Sepedophilus nigripennis	1.3209	0.1193	0.1379	0.6437	0.7018	1.2128	0.3765	0.3142	0.3235	0.3623	0.3520	0.3360	0.3120	m
Staphylinus dimidiaticornis	2.0023	0.1336	0.3150	0.6723	0.9770	1.4475	0.2829	0.3628	0.3380	0.2992	0.4366	0.2876	0.2758	m
Stenus brunnipes	1.2928	0.2819	0.1834	0.5786	0.9988	1.2829	0.1753	0.3712	0.3827	0.2461	0.4197	0.3514	0.2289	m
Stenus clavicornis	1.4376	0.3377	0.1781	0.6571	1.0527	1.2856	0.1788	0.3554	0.3434	0.3012	0.4028	0.3544	0.2428	m
Stenus pusillus	1.2007	0.3185	0.2506	0.5060	0.8764	1.3182	0.1984	0.3934	0.3607	0.2459	0.4333	0.3500	0.2167	m
Sunius propinquus	1.2867	0.1017	0.2386	0.9124	1.0283	0.9808	0.2734	0.3920	0.3375	0.2705	0.4038	0.3623	0.2339	m
Tachyporus nitidulus	1.3100	0.0966	0.1336	0.8066	0.6932	0.7656	0.3487	0.3208	0.3593	0.3199	0.3996	0.3215	0.2789	m
Tasgius morsitans	1.8111	0.1344	0.3783	0.9401	1.0894	1.4335	0.3097	0.3454	0.3288	0.3258	0.4410	0.3105	0.2485	m
Xantholinus elegans	1.6633	0.0876	0.2620	0.7564	1.4340	0.9857	0.3833	0.3842	0.3339	0.2820	0.4224	0.3273	0.2503	m
Xantholinus linearis	1.5461	0.0734	0.2431	0.7271	1.3635	1.0377	0.2673	0.3892	0.3331	0.2776	0.4300	0.3339	0.2360	m
Xantholinus longiventris	1.5846	0.0992	0.2201	0.6591	1.4011	0.9756	0.3013	0.3838	0.3586	0.2575	0.4235	0.3247	0.2518	m

Table A9.4.2 Capture data for beetles from pitfall traps at Canvey Wick experimental ground disturbance plots, 2015-2016. Three pitfall traps were open for 16 days each in May, July, and September. All plots were monitored before management in 2015 (n=27). Excavator disturbance and scraping treatments were applied in November 2015 (both n=9), and monitoring was repeated in 2016. Management treatments are coded C, unmanaged grassland in 2015, subsequently undisturbed; C[D] unmanaged grassland in 2015, plots for subsequent excavator disturbance; C[S] unmanaged grassland in 2015, plots for subsequent scraping; C, unmanaged grassland in 2016; D, plots after excavator disturbance; S, plots after scraping. Species were grouped as management indicators: stenotopic in early successional habitat; generalists associated with wetlands, woodland, or grassland; or eurytopic. Species also were grouped by trophic guild from adult feeding habits, as detritivorous, phytophagous, predatory, zoophagous, or unknown.

					2015			2016			
Family	Taxon	Authority	Management guild	Trophic group	C	C[D]	C[S]	C	D	S	Total
Anthicidae	*Anthicus antherinus*	(Linnaeus)	Stenotopic	Detritivorous	0	0	0	0	0	1	1
Anthicidae	*Notoxus monoceros*	(Linnaeus)	Stenotopic	Detritivorous	0	0	1	0	0	0	1
Apionidae	*Holotrichapion aethiops*	(Herbst)	Generalist	Phytophagous	0	0	0	0	1	0	1
Apionidae	*Ischnopterapion loti*	(Kirby)	Generalist	Phytophagous	0	2	2	4	6	1	15
Apionidae	*Ischnopterapion virens*	(Herbst)	Generalist	Phytophagous	0	0	0	1	0	0	1
Apionidae	*Oxystoma pomonae*	(Fabricius)	Generalist	Phytophagous	0	0	0	1	1	0	2
Apionidae	*Protapion assimile*	(Kirby)	Generalist	Phytophagous	0	1	0	0	0	0	1
Apionidae	*Protapion trifolii*	(Linnaeus)	Generalist	Phytophagous	0	0	0	0	0	1	1
Apionidae	*Stenopterapion tenue*	(Kirby)	Generalist	Phytophagous	0	0	0	1	0	0	1
Byrrhidae	*Byrrhus pilula*	(Linnaeus)	Generalist	Phytophagous	6	5	4	1	1	1	18
Byrrhidae	*Byrrhus pustulatus*	(Forster)	Generalist	Phytophagous	0	0	1	0	3	1	5
Byrrhidae	*Chaetophora spinosa*	(Rossi)	Generalist	Phytophagous	0	0	0	0	0	1	1
Byrrhidae	*Curimopsis maritima*	(Marsham)	Stenotopic	Phytophagous	0	0	0	2	1	1	4
Cantharidae	*Cantharis cryptica*	Ashe	Generalist	Predatory	0	0	1	0	0	0	1
Cantharidae	*Cantharis decipiens*	Baudi	Generalist	Predatory	0	0	1	0	0	0	1
Cantharidae	*Cantharis rufa*	Linnaeus	Generalist	Predatory	1	0	0	0	0	0	1
Cantharidae	*Rhagonycha fulva*	(Scopoli)	Eurytopic	Predatory	0	0	1	2	0	0	3
Carabidae	*Acupalpus dubius*	Schilsky	Generalist	Zoophagous	38	63	54	82	94	34	365
Carabidae	*Acupalpus parvulus*	(Sturm)	Generalist	Zoophagous	1	0	0	0	2	1	4

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Carabidae	*Agonum emarginatum*	(Gyllenhal)	Generalist	Zoophagous	149	138	131	157	77	24	676
Carabidae	*Agonum marginatum*	(Linnaeus)	Stenotopic	Zoophagous	0	0	0	0	0	4	4
Carabidae	*Amara aenea*	(De Geer)	Stenotopic	Phytophagous	11	9	13	5	2	1	41
Carabidae	*Amara bifrons*	(Gyllenhal)	Generalist	Phytophagous	1	0	0	0	0	0	1
Carabidae	*Amara communis*	(Panzer)	Generalist	Phytophagous	289	213	261	208	31	8	1010
Carabidae	*Amara convexior*	Stephens	Stenotopic	Phytophagous	27	22	20	6	3	2	80
Carabidae	*Amara eurynota*	(Panzer)	Stenotopic	Phytophagous	0	2	0	0	1	1	4
Carabidae	*Amara lunicollis*	Schiödte	Generalist	Phytophagous	692	710	762	328	50	7	2549
Carabidae	*Amara ovata*	(Fabricius)	Generalist	Phytophagous	1	0	3	1	0	1	6
Carabidae	*Amara tibialis*	(Paykull)	Stenotopic	Phytophagous	0	0	0	1	1	0	2
Carabidae	*Anisodactylus binotatus*	(Fabricius)	Generalist	Phytophagous	6	17	10	12	6	2	53
Carabidae	*Anthracus consputus*	(Duftschmid)	Stenotopic	Zoophagous	0	0	0	4	1	3	8
Carabidae	*Badister bullatus*	(Schrank)	Generalist	Zoophagous	1	2	1	1	0	1	6
Carabidae	*Bembidion assimile*	Gyllenhal	Generalist	Zoophagous	3	10	19	8	8	4	52
Carabidae	*Bembidion biguttatum*	(Fabricius)	Generalist	Zoophagous	0	0	4	0	1	1	6
Carabidae	*Bembidion lunulatum*	(Geoffroy in Fourcroy)	Stenotopic	Zoophagous	3	2	4	3	10	11	33
Carabidae	*Bembidion minimum*	(Fabricius)	Generalist	Zoophagous	0	0	0	0	0	1	1
Carabidae	*Bembidion properans*	(Stephens)	Generalist	Zoophagous	3	0	1	2	3	7	16
Carabidae	*Bembidion quadrimaculatum*	(Linnaeus)	Generalist	Zoophagous	1	1	3	2	2	74	83
Carabidae	*Bembidion varium*	(Olivier)	Stenotopic	Zoophagous	0	0	0	0	0	1	1
Carabidae	*Brachinus crepitans*	(Linnaeus)	Stenotopic	Zoophagous	0	2	3	1	1	5	12
Carabidae	*Bradycellus harpalinus*	(Audinet-Serville)	Eurytopic	Phytophagous	6	6	6	10	4	3	35
Carabidae	*Bradycellus verbasci*	(Duftschmid)	Generalist	Phytophagous	1	3	3	2	1	1	11
Carabidae	*Calathus ambiguus*	(Paykull)	Stenotopic	Zoophagous	0	0	1	0	0	0	1
Carabidae	*Calathus fuscipes*	(Goeze)	Generalist	Zoophagous	23	48	31	5	24	11	142
Carabidae	*Calathus melanocephalus*	(Linnaeus)	Generalist	Zoophagous	1	0	1	0	0	1	3
Carabidae	*Carabus nemoralis*	Müller, O.F.	Generalist	Zoophagous	12	5	6	17	7	1	48

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Carabidae	*Carabus violaceus*	Linnaeus	Generalist	Zoophagous	6	16	4	10	10	2	48
Carabidae	*Clivina fossor*	(Linnaeus)	Generalist	Zoophagous	32	41	26	72	42	16	229
Carabidae	*Curtonotus aulicus*	(Panzer)	Generalist	Phytophagous	0	1	1	0	0	3	5
Carabidae	*Curtonotus convexiusculus*	(Marsham)	Stenotopic	Phytophagous	0	2	0	0	0	0	2
Carabidae	*Demetrias atricapillus*	(Linnaeus)	Generalist	Zoophagous	0	0	0	1	0	0	1
Carabidae	*Dyschirius aeneus*	(Dejean)	Stenotopic	Zoophagous	0	0	0	0	3	19	22
Carabidae	*Dyschirius luedersi*	Wagner	Stenotopic	Zoophagous	0	0	0	0	1	0	1
Carabidae	*Dyschirius politus*	(Dejean)	Stenotopic	Zoophagous	0	0	0	0	0	16	16
Carabidae	*Elaphrus riparius*	(Linnaeus)	Stenotopic	Zoophagous	0	0	1	0	1	70	72
Carabidae	*Harpalus affinis*	(Schrank)	Generalist	Phytophagous	20	4	16	3	2	10	55
Carabidae	*Harpalus rubripes*	(Duftschmid)	Stenotopic	Phytophagous	0	0	0	1	4	1	6
Carabidae	*Harpalus rufipes*	(De Geer)	Eurytopic	Phytophagous	2	1	1	0	0	1	5
Carabidae	*Harpalus tardus*	(Panzer)	Stenotopic	Phytophagous	0	1	0	1	2	0	4
Carabidae	*Leistus ferrugineus*	(Linnaeus)	Generalist	Zoophagous	6	6	7	0	0	0	19
Carabidae	*Leistus fulvibarbis*	Dejean	Generalist	Zoophagous	0	1	0	0	0	1	2
Carabidae	*Loricera pilicornis*	(Fabricius)	Woodland	Zoophagous	0	0	1	0	0	0	1
Carabidae	*Microlestes maurus*	(Sturm)	Generalist	Zoophagous	2	2	2	1	0	2	9
Carabidae	*Nebria brevicollis*	(Fabricius)	Generalist	Zoophagous	4	7	4	2	16	8	41
Carabidae	*Notiophilus biguttatus*	(Fabricius)	Eurytopic	Zoophagous	1	0	0	1	1	0	3
Carabidae	*Notiophilus palustris*	(Duftschmid)	Generalist	Zoophagous	2	2	2	2	3	3	14
Carabidae	*Notiophilus substriatus*	Waterhouse, G.R.	Stenotopic	Zoophagous	2	0	2	0	1	2	7
Carabidae	*Ophonus puncticeps*	Stephens	Generalist	Phytophagous	0	0	1	0	0	0	1
Carabidae	*Ophonus rufibarbis*	(Fabricius)	Generalist	Phytophagous	0	0	0	1	0	0	1
Carabidae	*Panagaeus bipustulatus*	(Fabricius)	Stenotopic	Zoophagous	0	0	1	1	0	0	2
Carabidae	*Paradromius linearis*	(Olivier)	Stenotopic	Zoophagous	2	0	1	3	0	2	8
Carabidae	*Poecilus cupreus*	(Linnaeus)	Stenotopic	Zoophagous	47	48	38	25	52	11	221
Carabidae	*Pterostichus gracilis*	(Dejean)	Generalist	Zoophagous	6	6	8	20	16	14	70

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Carabidae	*Pterostichus melanarius*	(Illiger)	Generalist	Zoophagous	125	127	140	24	49	12	477
Carabidae	*Pterostichus niger*	(Schaller)	Generalist	Zoophagous	4	7	8	11	22	0	52
Carabidae	*Pterostichus nigrita*	(Paykull)	Generalist	Zoophagous	210	144	248	197	235	67	1101
Carabidae	*Pterostichus strenuus*	(Panzer)	Generalist	Zoophagous	56	46	47	42	11	2	204
Carabidae	*Pterostichus vernalis*	(Panzer)	Generalist	Zoophagous	1	1	1	2	0	0	5
Carabidae	*Stenolophus mixtus*	(Herbst)	Generalist	Phytophagous	1	1	3	8	5	0	18
Carabidae	*Stenolophus teutonus*	(Schrank)	Generalist	Phytophagous	0	0	0	2	8	2	12
Carabidae	*Stomis pumicatus*	(Panzer)	Generalist	Zoophagous	4	3	0	1	0	0	8
Carabidae	*Syntomus foveatus*	(Geoffroy in Fourcroy)	Stenotopic	Zoophagous	0	0	0	2	0	2	4
Carabidae	*Trechus obtusus*	Erichson	Eurytopic	Zoophagous	1	2	2	1	0	0	6
Carabidae	*Trechus quadristriatus*	(Schrank)	Eurytopic	Zoophagous	1	0	2	0	0	0	3
Cerambycidae	*Pseudovadonia livida*	(Fabricius)	Generalist	Phytophagous	0	1	0	0	0	0	1
Chrysomelidae	*Altica palustris*	Weise	Generalist	Phytophagous	0	0	0	0	2	0	2
Chrysomelidae	*Bruchus atomarius*	(Linnaeus)	Generalist	Phytophagous	0	1	0	0	0	0	1
Chrysomelidae	*Bruchus loti*	Paykull	Generalist	Phytophagous	0	0	0	1	0	0	1
Chrysomelidae	*Bruchus rufipes*	Herbst	Generalist	Phytophagous	0	0	0	2	0	0	2
Chrysomelidae	*Chaetocnema concinna*	(Marsham)	Eurytopic	Phytophagous	0	0	0	0	2	3	5
Chrysomelidae	*Chaetocnema hortensis*	(Fourcroy)	Eurytopic	Phytophagous	1	0	2	1	38	40	82
Chrysomelidae	*Longitarsus dorsalis*	(Fabricius)	Generalist	Phytophagous	1	0	1	0	0	0	2
Chrysomelidae	*Longitarsus parvulus*	(Paykull)	Eurytopic	Phytophagous	0	0	0	0	1	2	3
Chrysomelidae	*Longitarsus suturellus*	(Duftschmid)	Eurytopic	Phytophagous	0	9	1	12	6	3	31
Chrysomelidae	*Neocrepidodera ferruginea*	(Scopoli)	Eurytopic	Phytophagous	0	0	0	0	1	0	1
Chrysomelidae	*Podagrica fuscipes*	(Fabricius)	Eurytopic	Phytophagous	0	1	0	0	0	0	1
Clambidae	*Clambus pubescens*	Redtenbacher	Generalist	Unknown	3	1	4	16	15	4	43
Coccinellidae	*Coccinella septempunctata*	Linnaeus	Eurytopic	Predatory	2	0	0	0	1	9	12
Coccinellidae	*Harmonia axyridis*	(Pallas)	Eurytopic	Predatory	1	0	0	0	0	0	1
Coccinellidae	*Hippodamia variegata*	(Goeze)	Stenotopic	Predatory	0	0	0	0	0	9	9

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Coccinellidae	*Nephus redtenbacheri*	(Mulsant)	Generalist	Predatory	0	0	0	5	1	1	7
Coccinellidae	*Psyllobora vigintiduopunctata*	(Linnaeus)	Eurytopic	Predatory	1	0	1	0	0	0	2
Coccinellidae	*Rhyzobius litura*	(Fabricius)	Generalist	Predatory	1	1	0	5	1	1	9
Coccinellidae	*Subcoccinella vigintiquattuorpunctata*	(Linnaeus)	Generalist	Predatory	2	3	5	5	2	1	18
Coccinellidae	*Tytthaspis sedecimpunctata*	(Linnaeus)	Eurytopic	Predatory	3	1	1	16	11	1	33
Coccinellidae	*Tytthaspis sedecimpunctata*	(Linnaeus)	Generalist	Predatory	0	0	0	0	1	0	1
Corylophidae	*Corylophus sublaevipennis*	Jacquelin du Val	Generalist	Detritivorous	0	0	1	4	1	0	6
Cryptophagidae	*Atomaria testacea*	Stephens	Eurytopic	Detritivorous	0	0	0	0	0	1	1
Cryptophagidae	*Cryptophagus distinguendus*	Sturm	Eurytopic	Detritivorous	1	0	0	0	0	0	1
Cryptophagidae	*Ootypus globosus*	(Waltl)	Unknown	Detritivorous	0	0	0	2	1	0	3
Curculionidae	*Barypeithes pellucidus*	(Boheman)	Eurytopic	Phytophagous	0	1	0	3	3	2	9
Curculionidae	*Ceutorhynchus contractus*	(Marsham)	Eurytopic	Phytophagous	0	0	0	0	0	1	1
Curculionidae	*Euophryum confine*	(Broun)	Eurytopic	Phytophagous	0	0	0	0	1	0	1
Curculionidae	*Glocianus punctiger*	(Gyllenhal)	Generalist	Phytophagous	1	0	0	0	0	0	1
Curculionidae	*Hypera meles*	(Fabricius)	Generalist	Phytophagous	0	0	0	0	0	1	1
Curculionidae	*Hypera plantaginis*	(De Geer)	Generalist	Phytophagous	0	0	0	0	1	0	1
Curculionidae	*Otiorhynchus ovatus*	(Linnaeus)	Stenotopic	Phytophagous	0	1	0	0	0	0	1
Curculionidae	*Otiorhynchus rugosostriatus*	(Goeze)	Eurytopic	Phytophagous	1	0	0	0	0	0	1
Curculionidae	*Rhinoncus pericarpius*	(Linnaeus)	Eurytopic	Phytophagous	0	0	0	0	2	1	3
Curculionidae	*Rhinoncus perpendicularis*	(Reich)	Eurytopic	Phytophagous	0	0	1	0	0	0	1
Curculionidae	*Sitona cylindricollis*	(Fåhraeus)	Stenotopic	Phytophagous	0	0	0	0	1	0	1
Curculionidae	*Sitona humeralis*	Stephens	Generalist	Phytophagous	1	1	1	1	1	0	5
Curculionidae	*Sitona lepidus*	Gyllenhal	Eurytopic	Phytophagous	1	1	0	0	1	2	5
Curculionidae	*Sitona lineatus*	(Linnaeus)	Eurytopic	Phytophagous	6	8	1	4	7	2	28
Curculionidae	*Sitona sulcifrons*	(Thunberg)	Eurytopic	Phytophagous	0	0	0	0	0	1	1
Curculionidae	*Tanymecus palliatus*	(Fabricius)	Generalist	Phytophagous	0	0	1	0	1	0	2
Curculionidae	*Tychius junceus*	(Reich)	Generalist	Phytophagous	0	0	1	0	0	1	2

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Curculionidae	*Tychius picirostris*	(Fabricius)	Generalist	Phytophagous	0	1	0	0	0	0	1
Dytiscidae	*Agabus bipustulatus*	(Linnaeus)	Eurytopic	Predatory	0	1	0	0	0	0	1
Dytiscidae	*Agabus nebulosus*	(Forster)	Generalist	Predatory	0	0	0	1	1	0	2
Dytiscidae	*Hydroporus planus*	(Fabricius)	Generalist	Predatory	2	2	0	0	1	3	8
Elateridae	*Agriotes lineatus*	(Linnaeus)	Eurytopic	Phytophagous	137	117	138	178	220	37	827
Elateridae	*Agriotes sputator*	(Linnaeus)	Eurytopic	Phytophagous	0	0	0	2	1	1	4
Elateridae	*Agrypnus murinus*	(Linnaeus)	Generalist	Phytophagous	1	0	0	0	0	0	1
Erirhinidae	*Grypus equiseti*	(Fabricius)	Generalist	Phytophagous	0	0	0	0	0	1	1
Erirhinidae	*Notaris acridulus*	(Linnaeus)	Generalist	Phytophagous	0	0	0	0	1	0	1
Erirhinidae	*Notaris scirpi*	(Fabricius)	Generalist	Phytophagous	0	0	0	0	0	1	1
Erotylidae	*Cryptophilus integer*	(Heer)	Eurytopic	Unknown	0	0	0	0	0	1	1
Helophoridae	*Helophorus aequalis*	Thomson, C.G.	Generalist	Detritivorous	0	1	0	0	1	0	2
Helophoridae	*Helophorus alternans*	Gené	Generalist	Detritivorous	0	0	0	1	0	16	17
Helophoridae	*Helophorus brevipalpis*	Bedel	Generalist	Detritivorous	0	0	0	0	0	1	1
Helophoridae	*Helophorus grandis*	Illiger	Generalist	Detritivorous	0	0	0	0	0	1	1
Helophoridae	*Helophorus minutus*	Fabricius	Generalist	Detritivorous	1	1	1	2	1	2	8
Helophoridae	*Helophorus obscurus*	Mulsant	Generalist	Detritivorous	27	19	23	1	2	2	74
Histeridae	*Hister unicolor*	Linnaeus	Generalist	Predatory	2	9	3	33	10	0	57
Histeridae	*Kissister minimus*	(Aubé)	Generalist	Predatory	0	0	0	0	1	0	1
Histeridae	*Margarinotus purpurascens*	(Herbst)	Generalist	Predatory	0	0	0	1	3	0	4
Histeridae	*Saprinus aeneus*	(Fabricius)	Generalist	Predatory	0	0	0	1	0	0	1
Hydraenidae	*Ochthebius minimus*	(Fabricius)	Generalist	Detritivorous	0	0	0	0	0	5	5
Hydrophilidae	*Anacaena limbata*	(Fabricius)	Generalist	Detritivorous	1	2	3	0	0	1	7
Hydrophilidae	*Cercyon lateralis*	(Marsham)	Eurytopic	Detritivorous	0	0	0	1	0	0	1
Hydrophilidae	*Cercyon sternalis*	Sharp	Generalist	Detritivorous	0	1	0	0	0	0	1
Hydrophilidae	*Cercyon tristis*	(Illiger	Generalist	Detritivorous	0	1	0	1	0	0	2
Hydrophilidae	*Hydrobius fuscipes*	(Linnaeus)	Eurytopic	Detritivorous	0	1	0	0	3	0	4

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Hydrophilidae	*Megasternum concinnum*	(Marsham)	Eurytopic	Detritivorous	9	11	17	18	42	3	100
Hydrophilidae	*Sphaeridium bipustulatum*	Fabricius	Eurytopic	Detritivorous	0	0	0	1	0	0	1
Kateretidae	*Brachypterus glaber*	(Newman)	Eurytopic	Phytophagous	0	0	0	0	1	0	1
Latridiidae	*Corticarina minuta*	(Fabricius)	Eurytopic	Detritivorous	0	0	0	0	1	1	2
Latridiidae	*Enicmus transversus*	(Olivier)	Eurytopic	Detritivorous	0	0	0	1	1	1	3
Latridiidae	*Melanophthalma suturalis*	(Mannerheim)	Stenotopic	Detritivorous	0	0	0	0	0	1	1
Leiodidae	*Agathidium laevigatum*	Erichson	Generalist	Detritivorous	3	0	0	3	0	1	7
Leiodidae	*Catops fuscus*	(Panzer)	Generalist	Detritivorous	0	0	0	0	1	0	1
Leiodidae	*Catops grandicollis*	Erichson	Generalist	Detritivorous	2	2	1	0	1	0	6
Leiodidae	*Catops morio*	(Fabricius)	Generalist	Detritivorous	5	1	0	2	4	0	12
Leiodidae	*Catops nigricans*	(Spence)	Generalist	Detritivorous	2	0	0	4	0	0	6
Leiodidae	*Choleva agilis*	(Illiger)	Generalist	Detritivorous	0	1	0	4	1	0	6
Leiodidae	*Choleva oblonga*	Latreille	Generalist	Detritivorous	1	0	1	1	1	0	4
Leiodidae	*Leiodes rufipennis*	(Paykull)	Stenotopic	Detritivorous	0	5	0	1	1	3	10
Leiodidae	*Ptomaphagus subvillosus*	(Goeze)	Generalist	Detritivorous	0	1	0	0	0	0	1
Leiodidae	*Sciodrepoides watsoni*	(Spence)	Generalist	Detritivorous	1	0	0	2	1	1	5
Malachiidae	*Cordylepherus viridis*	(Fabricius)	Generalist	Predatory	3	3	2	0	1	0	9
Nitidulidae	*Meligethes aeneus*	(Fabricius)	Eurytopic	Phytophagous	0	0	0	0	0	1	1
Nitidulidae	*Meligethes carinulatus*	Förster	Generalist	Phytophagous	0	0	0	1	2	0	3
Nitidulidae	*Meligethes nigrescens*	Stephens	Eurytopic	Phytophagous	1	0	0	0	0	0	1
Oedemeridae	*Oedemera lurida*	(Marsham)	Eurytopic	Phytophagous	0	0	1	0	0	0	1
Ptiliidae	*Acrotrichis fascicularis*	(Herbst)	Generalist	Detritivorous	1	1	0	5	3	0	10
Scarabaeidae	*Aphodius plagiatus*	(Linnaeus)	Eurytopic	Detritivorous	1	1	0	0	11	14	27
Scarabaeidae	*Aphodius sphacelatus*	(Panzer)	Eurytopic	Detritivorous	0	0	1	2	1	0	4
Scarabaeidae	*Hoplia philanthus*	(Füessly)	Generalist	Detritivorous	1	0	0	0	0	0	1
Silphidae	*Nicrophorus vespillo*	(Linnaeus)	Eurytopic	Detritivorous	12	11	0	5	1	0	29
Silphidae	*Silpha laevigata*	Fabricius	Generalist	Predatory	0	0	0	0	1	1	2

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Silphidae	*Silpha tristis*	Illiger	Generalist	Predatory	185	159	128	211	207	24	914
Silphidae	*Thanatophilus sinuatus*	(Fabricius)	Generalist	Detritivorous	2	0	0	1	0	0	3
Staphylinidae	*Aleochara bipustulata*	(Linnaeus)	Eurytopic	Predatory	1	0	0	1	1	21	24
Staphylinidae	*Aleochara brevipennis*	Gravenhorst	Eurytopic	Predatory	17	15	12	99	528	42	713
Staphylinidae	*Aleochara curtula*	(Goeze)	Eurytopic	Predatory	10	3	1	3	4	0	21
Staphylinidae	*Aloconota gregaria*	(Erichson)	Eurytopic	Predatory	5	7	5	1	0	10	28
Staphylinidae	*Amischa analis*	(Gravenhorst)	Eurytopic	Predatory	14	19	14	25	61	17	150
Staphylinidae	*Amischa decipiens*	(Sharp)	Eurytopic	Predatory	0	3	2	1	1	2	9
Staphylinidae	*Anotylus insecatus*	(Gravenhorst)	Eurytopic	Detritivorous	0	0	0	0	1	0	1
Staphylinidae	*Anotylus nitidulus*	(Gravenhorst)	Eurytopic	Detritivorous	0	0	0	0	0	12	12
Staphylinidae	*Anotylus rugosus*	(Fabricius)	Eurytopic	Detritivorous	20	24	19	11	17	20	111
Staphylinidae	*Anotylus sculpturatus*	(Gravenhorst)	Eurytopic	Detritivorous	0	2	0	0	0	0	2
Staphylinidae	*Atheta aquatica*	(Thomson)	Eurytopic	Predatory	1	0	0	0	0	0	1
Staphylinidae	*Bledius gallicus*	(Gravenhorst)	Stenotopic	Phytophagous	0	1	0	1	1	143	146
Staphylinidae	*Bledius opacus*	(Block)	Stenotopic	Phytophagous	0	0	0	0	0	1	1
Staphylinidae	*Brachygluta fossulata*	(Reichenbach)	Generalist	Predatory	6	2	2	12	14	4	40
Staphylinidae	*Callicerus obscurus*	Gravenhorst	Generalist	Predatory	1	0	0	0	0	0	1
Staphylinidae	*Callicerus rigidicornis*	(Erichson)	Generalist	Predatory	3	5	3	1	2	0	14
Staphylinidae	*Calodera protensa*	Mannerheim	Generalist	Predatory	14	13	19	89	82	98	315
Staphylinidae	*Carpelimus corticinus*	(Gravenhorst)	Stenotopic	Detritivorous	1	1	2	0	8	83	95
Staphylinidae	*Carpelimus erichsoni*	(Sharp)	Stenotopic	Detritivorous	0	0	0	0	2	12	14
Staphylinidae	*Carpelimus gracilis*	(Mannerheim)	Stenotopic	Detritivorous	0	0	0	0	2	79	81
Staphylinidae	*Carpelimus incongruus*	Steel	Stenotopic	Detritivorous	0	0	0	0	0	4	4
Staphylinidae	*Carpelimus rivularis*	(Motschulsky)	Stenotopic	Detritivorous	0	0	0	0	0	1	1
Staphylinidae	*Cypha longicornis*	(Paykull)	Eurytopic	Predatory	0	0	0	1	1	0	2
Staphylinidae	*Dimetrota atramentaria*	(Gyllenhal)	Eurytopic	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Dinaraea angustula*	(Gyllenhal)	Eurytopic	Predatory	2	1	1	14	21	46	85

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Staphylinidae	*Drusilla canaliculata*	(Fabricius)	Eurytopic	Predatory	412	555	391	541	215	31	2145
Staphylinidae	*Gabrius appendiculatus*	Sharp	Generalist	Predatory	1	0	0	0	0	0	1
Staphylinidae	*Gabrius breviventer*	(Sperk)	Generalist	Predatory	22	35	28	9	38	6	138
Staphylinidae	*Gyrohypnus wagneri*	(Scheerpeltz)	Unknown	Predatory	1	0	0	0	1	0	2
Staphylinidae	*Ilyobates bennetti*	Donisthorpe	Generalist	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Ischnosoma splendidum*	(Gravenhorst)	Generalist	Predatory	0	0	0	1	4	1	6
Staphylinidae	*Lamprinodes saginatus*	(Gravenhorst)	Generalist	Predatory	0	0	0	6	2	3	11
Staphylinidae	*Lathrobium brunnipes*	(Fabricius)	Generalist	Predatory	3	2	0	2	0	1	8
Staphylinidae	*Lathrobium fulvipenne*	(Gravenhorst)	Generalist	Predatory	1	1	1	0	0	0	3
Staphylinidae	*Lathrobium geminum*	Kraatz	Generalist	Predatory	6	11	15	12	18	9	71
Staphylinidae	*Liogluta longiuscula*	(Gravenhorst)	Eurytopic	Predatory	0	0	0	0	5	0	5
Staphylinidae	*Liogluta microptera*	Thomson, C.G.	Eurytopic	Predatory	0	0	0	0	1	0	1
Staphylinidae	*Liogluta pagana*	(Erichson)	Generalist	Predatory	0	0	0	0	1	0	1
Staphylinidae	*Lobrathium multipunctum*	(Gravenhorst)	Generalist	Predatory	0	0	0	0	0	3	3
Staphylinidae	*Meotica exilis*	(Knoch)	Generalist	Predatory	0	1	1	2	1	1	6
Staphylinidae	*Microdota aegra*	(Heer)	Unknown	Predatory	0	0	1	0	0	0	1
Staphylinidae	*Mycetoporus*	Lundström	Unknown	Predatory	0	0	0	0	1	0	1
Staphylinidae	*Neobisnius procerulus*	(Gravenhorst)	Stenotopic	Predatory	0	0	0	1	2	45	48
Staphylinidae	*Ochthephilum collare*	(Reitter)	Generalist	Predatory	4	1	0	0	1	0	6
Staphylinidae	*Ocypus aeneocephalus*	(De Geer)	Generalist	Predatory	0	2	4	1	0	0	7
Staphylinidae	*Ocypus fuscatus*	(Gravenhorst)	Generalist	Predatory	10	6	11	7	1	6	41
Staphylinidae	*Ocypus olens*	(Müller, O.F.)	Eurytopic	Predatory	38	40	44	20	10	4	156
Staphylinidae	*Oligota pumilio*	Kiesenwetter	Eurytopic	Predatory	0	0	0	0	1	0	1
Staphylinidae	*Ontholestes murinus*	(Linnaeus)	Eurytopic	Predatory	0	0	1	0	1	0	2
Staphylinidae	*Othius laeviusculus*	Stephens	Eurytopic	Predatory	0	0	0	2	0	0	2
Staphylinidae	*Othius punctulatus*	(Goeze)	Generalist	Predatory	1	1	2	2	0	0	6
Staphylinidae	*Oxypoda brevicornis*	(Stephens)	Eurytopic	Predatory	0	0	0	0	0	1	1

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Staphylinidae	*Oxypoda elongatula*	Aubé	Generalist	Predatory	0	3	1	0	0	0	4
Staphylinidae	*Oxypoda procerula*	Mannerheim	Generalist	Predatory	1	2	0	2	0	0	5
Staphylinidae	*Paederus littoralis*	Gravenhorst	Stenotopic	Predatory	3	4	5	3	13	2	30
Staphylinidae	*Pella limbata*	(Paykull)	Generalist	Predatory	2	1	4	70	6	1	84
Staphylinidae	*Philhygra elongatula*	(Gravenhorst)	Generalist	Predatory	11	21	11	8	15	19	85
Staphylinidae	*Philhygra volans*	(Scriba)	Generalist	Predatory	0	0	0	0	0	2	2
Staphylinidae	*Philonthus carbonarius*	(Gravenhorst)	Generalist	Predatory	0	0	0	1	2	0	3
Staphylinidae	*Philonthus cognatus*	Stephens	Eurytopic	Predatory	2	2	3	3	1	1	12
Staphylinidae	*Philonthus cruentatus*	(Gmelin in Linnaeus)	Eurytopic	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Philonthus debilis*	(Gravenhorst)	Eurytopic	Predatory	1	0	0	0	0	0	1
Staphylinidae	*Philonthus marginatus*	(Müller, O.F.)	Generalist	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Philonthus micans*	(Gravenhorst)	Generalist	Predatory	0	0	0	0	1	0	1
Staphylinidae	*Philonthus quisquiliarius*	(Gyllenhal)	Stenotopic	Predatory	0	0	0	0	1	7	8
Staphylinidae	*Philonthus succicola*	Thomson, C.G.	Generalist	Predatory	1	0	1	0	0	0	2
Staphylinidae	*Philonthus varians*	(Paykull)	Eurytopic	Predatory	1	0	0	0	0	0	1
Staphylinidae	*Platydracus stercorarius*	(Olivier)	Generalist	Predatory	14	10	7	4	4	1	40
Staphylinidae	*Platystethus alutaceus*	Thomson, C.G.	Stenotopic	Predatory	0	0	0	0	1	3	4
Staphylinidae	*Platystethus nitens*	(Sahlberg)	Stenotopic	Predatory	0	0	0	0	0	7	7
Staphylinidae	*Quedius curtipennis*	Bernhauer	Generalist	Predatory	5	2	9	4	1	0	21
Staphylinidae	*Quedius levicollis*	(Brullé)	Generalist	Predatory	3	7	4	5	2	0	21
Staphylinidae	*Quedius maurorufus*	(Gravenhorst)	Generalist	Predatory	0	0	0	1	2	0	3
Staphylinidae	*Quedius molochinus*	(Gravenhorst)	Eurytopic	Predatory	24	15	18	0	0	0	57
Staphylinidae	*Quedius nemoralis*	Baudi	Eurytopic	Predatory	0	1	2	0	0	0	3
Staphylinidae	*Quedius nigriceps*	Kraatz	Generalist	Predatory	0	0	1	0	0	0	1
Staphylinidae	*Quedius persimilis*	Mulsant & Rey	Eurytopic	Predatory	0	1	0	0	0	0	1
Staphylinidae	*Quedius picipes*	(Mannerheim)	Eurytopic	Predatory	1	2	4	13	10	0	30
Staphylinidae	*Quedius schatzmayri*	Gridelli	Eurytopic	Predatory	1	1	2	0	0	0	4

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Staphylinidae	*Quedius semiaeneus*	(Stephens)	Eurytopic	Predatory	0	0	0	2	0	0	2
Staphylinidae	*Quedius semiobscurus*	(Marsham)	Eurytopic	Predatory	3	7	4	22	19	0	55
Staphylinidae	*Rugilus erichsonii*	(Fauvel)	Generalist	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Rugilus rufipes*	Germar	Generalist	Predatory	0	1	2	1	1	0	5
Staphylinidae	*Sepedophilus immaculatus*	(Stephens)	Generalist	Predatory	0	1	0	1	0	0	2
Staphylinidae	*Sepedophilus marshami*	(Stephens)	Eurytopic	Predatory	1	0	2	1	0	0	4
Staphylinidae	*Sepedophilus nigripennis*	(Stephens)	Generalist	Predatory	0	0	0	1	1	2	4
Staphylinidae	*Staphylinus dimidiaticornis*	Gemminger	Eurytopic	Predatory	38	33	47	84	80	2	284
Staphylinidae	*Stenichnus collaris*	(Müller, P.W.J. & Kunze)	Unknown	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Stenichnus scutellaris*	(Müller, P.W.J. & Kunze)	Unknown	Predatory	0	0	0	1	1	0	2
Staphylinidae	*Stenus bimaculatus*	Gyllenhal	Generalist	Predatory	0	0	0	1	0	1	2
Staphylinidae	*Stenus brunnipes*	Stephens	Eurytopic	Predatory	0	1	0	11	5	1	18
Staphylinidae	*Stenus clavicornis*	(Scopoli)	Eurytopic	Predatory	9	10	8	42	24	8	101
Staphylinidae	*Stenus fulvicornis*	Stephens	Generalist	Predatory	3	1	4	2	2	0	12
Staphylinidae	*Stenus juno*	(Paykull)	Generalist	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Stenus nanus*	Stephens	Eurytopic	Predatory	0	0	0	0	0	2	2
Staphylinidae	*Stenus ossium*	Stephens	Generalist	Predatory	1	0	2	0	0	0	3
Staphylinidae	*Stenus pusillus*	Stephens	Eurytopic	Predatory	6	1	1	3	9	2	22
Staphylinidae	*Sunius melanocephalus*	(Fabricius)	Eurytopic	Predatory	0	0	0	0	2	0	2
Staphylinidae	*Sunius propinquus*	(Brisout de Barneville)	Stenotopic	Predatory	0	0	0	1	1	2	4
Staphylinidae	*Tachinus flavolimbatus*	Pandellé	Generalist	Predatory	1	0	0	0	0	0	1
Staphylinidae	*Tachinus laticollis*	Gravenhorst	Generalist	Predatory	0	0	0	2	0	0	2
Staphylinidae	*Tachinus rufipes*	(Linnaeus)	Generalist	Predatory	7	5	10	2	0	0	24
Staphylinidae	*Tachyporus atriceps*	Stephens	Generalist	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Tachyporus chrysomelinus*	(Linnaeus)	Generalist	Predatory	0	1	0	0	0	0	1
Staphylinidae	*Tachyporus dispar*	(Paykull)	Eurytopic	Predatory	2	0	3	1	0	1	7
Staphylinidae	*Tachyporus hypnorum*	(Fabricius)	Eurytopic	Predatory	2	0	5	0	2	0	9

					2015			2016			
Family	Taxon	Authority	Management guild	Trophic group	C	C[D]	C[S]	C	D	S	Total
Staphylinidae	*Tachyporus nitidulus*	(Fabricius)	Eurytopic	Predatory	1	1	1	1	0	7	11
Staphylinidae	*Tasgius globulifer*	(Geoffroy)	Stenotopic	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Tasgius melanarius*	(Heer)	Generalist	Predatory	0	0	0	0	2	0	2
Staphylinidae	*Tasgius morsitans*	(Rossi)	Generalist	Predatory	4	0	3	3	1	1	12
Staphylinidae	*Tasgius winkleri*	Bernhauer	Stenotopic	Predatory	9	5	6	1	1	0	22
Staphylinidae	*Thecturota marchii*	(Dodero)	Stenotopic	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Thinonoma atra*	(Gravenhorst)	Stenotopic	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Tinotus morion*	(Gravenhorst)	Eurytopic	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Xantholinus elegans*	(Olivier)	Stenotopic	Predatory	4	2	2	7	5	2	22
Staphylinidae	*Xantholinus linearis*	(Olivier)	Eurytopic	Predatory	2	1	3	9	5	1	21
Staphylinidae	*Xantholinus longiventris*	Heer	Eurytopic	Predatory	61	59	58	106	156	95	535
Tenebrionidae	*Helops caeruleus*	(Linnaeus)	Unknown	Detritivorous	0	0	1	0	0	0	1
Throscidae	*Trixagus obtusus*	(Curtis)	Generalist	Unknown	8	4	6	0	2	0	20

Table A9.4.3 Capture data for spiders from pitfall traps at Canvey Wick experimental ground disturbance plots, 2015-2016. Three pitfall traps were open for 16 days each in May, July, and September. All plots were monitored before management in 2015 (n=27). Excavator disturbance and scraping treatments were applied in November 2015 (both n=9), and monitoring was repeated in 2016. Management treatments are coded C, unmanaged grassland in 2015, subsequently undisturbed; C[D] unmanaged grassland in 2015, plots for subsequent excavator disturbance; C[S] unmanaged grassland in 2015, plots for subsequent scraping; C, unmanaged grassland in 2016; D, plots after excavator disturbance; S, plots after scraping. Species were grouped as management indicators: stenotopic in early successional habitat; generalists associated with wetlands, woodland, or grassland; or eurytopic. Species also were grouped by trophic guild (all spiders predatory).

					2015			2016			
Family	Taxon	Authority	Management guild	Trophic group	C	C[D]	C[S]	C	D	S	Total
Agelenidae	*Agelena labyrinthica*	(Clerck)	Generalist	Predatory	0	1	0	0	0	0	1
Araneidae	*Hypsosinga pygmaea*	(Sundevall)	Generalist	Predatory	1	1	0	0	0	0	2
Clubionidae	*Clubiona diversa*	O.P.-Cambridge	Generalist	Predatory	0	2	4	0	0	1	7
Clubionidae	*Clubiona neglecta*	O.P.-Cambridge	Stenotopic	Predatory	0	1	2	0	0	0	3
Clubionidae	*Clubiona reclusa*	O.P.-Cambridge	Eurytopic	Predatory	1	1	0	0	0	0	2
Dictynidae	*Argenna subnigra*	(O.P.-Cambridge)	Stenotopic	Predatory	0	0	0	0	0	1	1
Dictynidae	*Dictyna latens*	(Fabricius)	Generalist	Predatory	0	0	1	0	0	0	1
Dysderidae	*Dysdera crocata*	C.L.Koch	Stenotopic	Predatory	0	0	1	0	1	0	2
Gnaphosidae	*Drassodes cupreus*	(Blackwall)	Stenotopic	Predatory	6	7	3	0	1	5	22
Gnaphosidae	*Drassodes pubescens*	(Thorell)	Generalist	Predatory	0	0	0	0	0	2	2
Gnaphosidae	*Drassyllus pusillus*	(C.L.Koch)	Stenotopic	Predatory	41	18	22	108	43	19	251
Gnaphosidae	*Haplodrassus signifer*	(C.L.Koch)	Stenotopic	Predatory	3	2	6	14	16	9	50
Gnaphosidae	*Micaria pulicaria*	(Sundevall)	Eurytopic	Predatory	5	7	0	6	19	16	53
Gnaphosidae	*Trachyzelotes pedestris*	(C.L.Koch)	Stenotopic	Predatory	3	2	4	5	2	4	20
Gnaphosidae	*Zelotes apricorum*	(L.Koch)	Stenotopic	Predatory	5	7	10	5	1	2	30
Gnaphosidae	*Zelotes latreillei*	(Simon)	Stenotopic	Predatory	8	8	8	9	7	1	41
Hahniidae	*Antistea elegans*	(Blackwall)	Generalist	Predatory	3	3	6	1	1	0	14
Hahniidae	*Hahnia nava*	(Blackwall)	Eurytopic	Predatory	37	12	18	343	160	45	615
Linyphiidae	*Bathyphantes gracilis*	(Blackwall)	Eurytopic	Predatory	1	2	1	6	20	2	32
Linyphiidae	*Bathyphantes parvulus*	(Westring)	Eurytopic	Predatory	2	2	4	0	1	5	14

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Linyphiidae	*Centromerita bicolor*	(Blackwall)	Eurytopic	Predatory	0	1	1	0	0	0	2
Linyphiidae	*Centromerita concinna*	(Thorell)	Generalist	Predatory	0	0	0	0	0	3	3
Linyphiidae	*Centromerus sylvaticus*	(Blackwall)	Generalist	Predatory	0	1	0	0	0	0	1
Linyphiidae	*Ceratinella brevipes*	(Westring)	Generalist	Predatory	0	0	0	0	1	0	1
Linyphiidae	*Ceratinopsis stativa*	(Simon)	Generalist	Predatory	0	0	0	2	1	0	3
Linyphiidae	*Dicymbium*	(Linnaeus)	Eurytopic	Predatory	1	2	7	6	5	0	21
Linyphiidae	*Dicymbium nigrum*	(Blackwall)	Eurytopic	Predatory	0	0	1	0	0	0	1
Linyphiidae	*Erigone arctica*	(White)	Stenotopic	Predatory	0	0	0	0	0	1	1
Linyphiidae	*Erigone atra*	Blackwall	Eurytopic	Predatory	0	0	2	9	45	124	180
Linyphiidae	*Erigone dentipalpis*	(Wider)	Eurytopic	Predatory	0	1	0	4	14	138	157
Linyphiidae	*Gongylidiellum vivum*	(O.P.-Cambridge)	Generalist	Predatory	0	3	2	11	6	0	22
Linyphiidae	*Meioneta beata*	(O.P.-Cambridge)	Generalist	Predatory	4	0	2	46	20	3	75
Linyphiidae	*Meioneta rurestris*	(C.L.Koch)	Eurytopic	Predatory	0	0	0	2	1	0	3
Linyphiidae	*Mermessus trilobatus*	(Emerton)	Stenotopic	Predatory	0	0	0	0	1	0	1
Linyphiidae	*Micrargus subaequalis*	(Westring)	Generalist	Predatory	1	0	0	0	0	0	1
Linyphiidae	*Microlinyphia pusilla*	(Sundevall)	Generalist	Predatory	0	0	0	1	0	1	2
Linyphiidae	*Microneta viaria*	(Blackwall)	Generalist	Predatory	2	0	0	1	0	0	3
Linyphiidae	*Oedothorax apicatus*	(Blackwall)	Eurytopic	Predatory	1	5	1	1	9	20	37
Linyphiidae	*Oedothorax fuscus*	(Blackwall)	Eurytopic	Predatory	1	0	2	1	28	25	57
Linyphiidae	*Oedothorax gibbosus*	(Blackwall)	Generalist	Predatory	0	0	0	3	6	10	19
Linyphiidae	*Oedothorax retusus*	(Westring)	Eurytopic	Predatory	5	4	8	7	38	54	116
Linyphiidae	*Palliduphantes ericaeus*	(Blackwall)	Generalist	Predatory	0	0	0	1	0	0	1
Linyphiidae	*Panamomops sulcifrons*	(Wider)	Generalist	Predatory	0	1	1	8	4	2	16
Linyphiidae	*Pelecopsis parallela*	(Wider)	Generalist	Predatory	1	3	0	3	2	0	9
Linyphiidae	*Pocadicnemis juncea*	Locket & Millidge	Eurytopic	Predatory	1	0	1	18	8	0	28
Linyphiidae	*Porrhomma pygmaeum*	(Blackwall)	Generalist	Predatory	0	0	0	0	1	1	2
Linyphiidae	*Prinerigone vagans*	(Audouin)	Eurytopic	Predatory	0	0	0	0	1	11	12

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Linyphiidae	*Tapinocyba praecox*	(O.P.-Cambridge)	Eurytopic	Predatory	0	0	0	1	0	0	1
Linyphiidae	*Tenuiphantes tenuis*	(Blackwall)	Eurytopic	Predatory	5	8	6	36	27	13	95
Linyphiidae	*Tiso vagans*	(Blackwall)	Eurytopic	Predatory	0	0	1	1	0	0	2
Linyphiidae	*Walckenaeria acuminata*	Blackwall	Eurytopic	Predatory	0	1	1	0	0	0	2
Linyphiidae	*Walckenaeria antica*	(Wider)	Eurytopic	Predatory	0	0	0	0	2	1	3
Linyphiidae	*Walckenaeria atrotibialis*	(O.P.-Cambridge)	Generalist	Predatory	2	2	5	2	0	0	11
Linyphiidae	*Walckenaeria nudipalpis*	(Westring)	Eurytopic	Predatory	0	0	0	0	1	1	2
Liocranidae	*Agraecina striata*	(Kulczynski)	Generalist	Predatory	10	11	5	79	28	2	135
Liocranidae	*Agroeca proxima*	(O.P.-Cambridge)	Generalist	Predatory	4	2	6	0	0	0	12
Liocranidae	*Phrurolithus festivus*	(C.L.Koch)	Eurytopic	Predatory	3	3	1	4	2	4	17
Lycosidae	*Alopecosa pulverulenta*	(Clerck)	Eurytopic	Predatory	320	378	422	253	186	23	1582
Lycosidae	*Arctosa leopardus*	(Sundevall)	Stenotopic	Predatory	1	0	1	10	61	105	178
Lycosidae	*Pardosa agrestis*	(Westring)	Stenotopic	Predatory	0	0	0	0	1	0	1
Lycosidae	*Pardosa nigriceps*	(Thorell)	Eurytopic	Predatory	33	26	36	7	13	13	128
Lycosidae	*Pardosa palustris*	(Linnaeus)	Generalist	Predatory	1	0	2	1	9	1	14
Lycosidae	*Pardosa prativaga*	(L.Koch)	Eurytopic	Predatory	52	45	65	49	171	34	416
Lycosidae	*Pardosa pullata*	(Clerck)	Generalist	Predatory	463	345	466	328	529	103	2234
Lycosidae	*Pirata piraticus*	(Clerck)	Generalist	Predatory	0	1	0	2	2	1	6
Lycosidae	*Pirata uliginosus*	(Thorell)	Generalist	Predatory	1	0	0	1	0	0	2
Lycosidae	*Trochosa ruricola*	(De Geer)	Eurytopic	Predatory	66	58	83	49	121	46	423
Lycosidae	*Trochosa terricola*	Thorell	Eurytopic	Predatory	136	99	126	25	30	6	422
Mimetidae	*Ero cambridgei*	Kulczynski	Woodland	Predatory	0	0	1	1	0	0	2
Philodromidae	*Thanatus striatus*	C.L.Koch	Eurytopic	Predatory	1	1	1	1	0	2	6
Philodromidae	*Tibellus oblongus*	(Walckenaer)	Generalist	Predatory	4	1	0	0	0	0	5
Pisauridae	*Pisaura mirabilis*	(Clerck)	Eurytopic	Predatory	0	0	0	1	0	0	1
Salticidae	*Euophrys frontalis*	(Walckenaer)	Eurytopic	Predatory	1	1	1	2	0	1	6
Salticidae	*Heliophanus cupreus*	(Walckenaer)	Eurytopic	Predatory	0	0	0	0	1	0	1

Family	Taxon	Authority	Management guild	Trophic group	2015			2016			Total
					C	C[D]	C[S]	C	D	S	
Salticidae	*Heliophanus flavipes*	(Hahn)	Eurytopic	Predatory	0	0	1	0	1	1	3
Salticidae	*Sibianor aurocinctus*	(Ohlert)	Stenotopic	Predatory	0	1	0	1	2	0	4
Tetragnathidae	*Pachygnatha degeeri*	Sundevall	Eurytopic	Predatory	16	19	15	14	62	10	136
Theridiidae	*Enoplognatha thoracica*	(Hahn)	Eurytopic	Predatory	0	0	0	0	0	2	2
Theridiidae	*Episinus angulatus*	(Blackwall)	Generalist	Predatory	1	0	0	0	0	0	1
Theridiidae	*Robertus arundineti*	(O.P.-Cambridge)	Generalist	Predatory	1	1	2	0	0	4	8
Theridiidae	*Robertus lividus*	(Blackwall)	Generalist	Predatory	0	0	0	0	0	1	1
Thomisidae	*Ozyptila brevipes*	(Hahn)	Generalist	Predatory	0	1	0	0	1	0	2
Thomisidae	*Ozyptila sanctuaria*	(O.P.-Cambridge)	Generalist	Predatory	11	11	9	0	2	0	33
Thomisidae	*Ozyptila simplex*	(O.P.-Cambridge)	Stenotopic	Predatory	25	17	19	89	107	12	269
Thomisidae	*Pachygnatha degeeri*	Turner	Eurytopic	Predatory	0	0	0	0	1	0	1
Thomisidae	*Xysticus cristatus*	(Clerck)	Eurytopic	Predatory	2	0	3	0	8	7	20
Thomisidae	*Xysticus kochi*	Thorell	Stenotopic	Predatory	1	0	2	1	11	10	25
Zodariidae	*Zodarion italicum*	(Canestrini)	Stenotopic	Predatory	1	2	2	2	3	0	10
Zoridae	*Zora spinimana*	(Sundevall)	Generalist	Predatory	19	25	17	8	0	1	70

Appendix 9.5

Table A9.5.1 Capture data for beetles from aggregate mounds experiment at Hadleigh Park, Pitsea Landfill, and Rainham Marshes, in 2015 and 2016. Treatments were 5-tonne mounds of recycled bulk fill (n=9) or Thanet sand (n=9), and unmanaged grassland (n=9). There were two pitfall traps per monitoring plot in each year, open for 16 days in May, July, and September. Plus symbol (+) by species name denotes status as a stenotopic indicator of early successional habitat in this study. Species were grouped by trophic guild from adult feeding habits, as detritivorous, phytophagous, predatory, zoophagous, or unknown.

				2015			2016			
Family	Taxon	Authority	Trophic group	Control	RBF	Thanet	Control	RBF	Thanet	Total
Alexiidae	*Sphaerosoma pilosum*	(Panzer)	Detritivorous	1	0	0	0	0	0	1
Anthicidae	*Anthicus antherinus*+	(Linnaeus)	Detritivorous	0	13	0	0	6	2	21
Apionidae	*Aspidapion radiolus*	(Marsham)	Phytophagous	0	0	0	0	1	0	1
Apionidae	*Ceratapion carduorum*	(Kirby)	Phytophagous	1	4	2	2	23	9	41
Apionidae	*Ceratapion gibbirostre*	(Gyllenhal)	Phytophagous	0	0	0	0	0	1	1
Apionidae	*Ceratapion onopordi*	(Kirby)	Phytophagous	1	2	3	0	7	24	37
Apionidae	*Malvapion malvae*	(Fabricius)	Phytophagous	0	0	0	0	0	1	1
Apionidae	*Omphalapion hookerorum*+	(Kirby)	Phytophagous	0	0	0	0	1	0	1
Apionidae	*Protapion assimile*	(Kirby)	Phytophagous	0	0	0	0	1	0	1
Apionidae	*Protapion trifolii*	(Linnaeus)	Phytophagous	0	0	0	0	0	1	1
Apionidae	*Stenopterapion tenue*	(Kirby)	Phytophagous	0	1	0	0	0	0	1
Apionidae	*Taeniapion urticarium*	(Herbst)	Phytophagous	0	0	0	1	0	1	2
Byrrhidae	*Byrrhus pilula*	(Linnaeus)	Phytophagous	5	2	1	1	0	1	10
Byrrhidae	*Simplocaria semistriata*	(Fabricius)	Phytophagous	0	0	1	0	0	1	2
Cantharidae	*Cantharis decipiens*	Baudi	Predatory	0	1	0	0	0	0	1
Cantharidae	*Cantharis lateralis*	Linnaeus	Predatory	0	4	0	0	0	0	4
Cantharidae	*Cantharis pallida*	Goeze	Predatory	0	0	0	0	0	1	1
Cantharidae	*Cantharis rufa*	Linnaeus	Predatory	0	0	0	0	0	2	2
Cantharidae	*Cantharis rustica*	Fallén	Predatory	0	2	0	0	0	1	3
Cantharidae	*Rhagonycha fulva*	(Scopoli)	Predatory	0	2	0	0	2	10	14

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Cantharidae	*Rhagonycha limbata*	Thomson	Predatory	0	0	0	0	1	0	1
Carabidae	*Acupalpus dubius*	Schilsky	Predatory	0	1	0	1	2	2	6
Carabidae	*Acupalpus meridianus*	(Linnaeus)	Predatory	0	0	1	0	0	0	1
Carabidae	*Agonum fuliginosum*	(Panzer)	Predatory	1	0	0	0	0	0	1
Carabidae	*Amara aenea*+	(De Geer)	Phytophagous	3	1	2	2	1	3	12
Carabidae	*Amara communis*	(Panzer)	Phytophagous	3	0	0	0	0	1	4
Carabidae	*Amara convexior*+	Stephens	Phytophagous	4	6	5	3	12	25	55
Carabidae	*Amara eurynota*+	(Panzer)	Phytophagous	3	3	1	1	100	6	114
Carabidae	*Amara familiaris*+	(Duftschmid)	Phytophagous	2	0	3	0	0	1	6
Carabidae	*Amara lunicollis*	Schiödte	Phytophagous	23	6	4	29	4	19	85
Carabidae	*Amara ovata*	(Fabricius)	Phytophagous	1	0	1	1	10	2	15
Carabidae	*Amara plebeja*	(Gyllenhal)	Phytophagous	0	0	0	0	1	0	1
Carabidae	*Amara similata*	(Gyllenhal)	Phytophagous	29	2	2	0	5	1	39
Carabidae	*Amara tibialis*+	(Paykull)	Phytophagous	1	0	0	1	0	0	2
Carabidae	*Anchomenus dorsalis*	(Pontoppidan)	Zoophagous	5	6	3	9	7	2	32
Carabidae	*Anisodactylus binotatus*	(Fabricius)	Zoophagous	4	0	0	0	0	0	4
Carabidae	*Badister bullatus*	(Schrank)	Zoophagous	14	5	0	15	2	8	44
Carabidae	*Bembidion biguttatum*	(Fabricius)	Zoophagous	0	0	0	3	0	2	5
Carabidae	*Bembidion femoratum*+	Sturm	Zoophagous	0	0	3	0	0	0	3
Carabidae	*Bembidion lampros*	(Herbst)	Zoophagous	5	2	6	0	0	0	13
Carabidae	*Bembidion lunulatum*+	(Geoffroy in Fourcroy)	Zoophagous	7	0	0	1	0	3	11
Carabidae	*Bembidion minimum*	(Fabricius)	Zoophagous	2	0	1	1	1	0	5
Carabidae	*Bembidion obtusum*	Audinet-Serville	Zoophagous	37	27	11	78	69	41	263
Carabidae	*Bembidion properans*	(Stephens)	Zoophagous	3	0	2	1	0	1	7
Carabidae	*Bembidion quadrimaculatum*	(Linnaeus)	Zoophagous	1	1	8	0	1	0	11
Carabidae	*Brachinus crepitans*+	(Linnaeus)	Zoophagous	13	20	6	14	18	3	74
Carabidae	*Bradycellus verbasci*	(Duftschmid)	Phytophagous	0	0	2	0	6	5	13

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Carabidae	*Calathus ambiguus*+	(Paykull)	Zoophagous	0	0	1	0	0	0	1
Carabidae	*Calathus fuscipes*	(Goeze)	Zoophagous	2	1	2	4	1	3	13
Carabidae	*Calathus melanocephalus*	(Linnaeus)	Zoophagous	6	0	0	2	2	2	12
Carabidae	*Carabus nemoralis*	Müller, O.F.	Zoophagous	2	0	0	0	4	8	14
Carabidae	*Carabus violaceus*	Linnaeus	Zoophagous	0	0	0	0	1	1	2
Carabidae	*Clivina fossor*	(Linnaeus)	Zoophagous	1	0	0	0	0	0	1
Carabidae	*Curtonotus aulicus*	(Panzer)	Phytophagous	0	4	28	7	45	62	146
Carabidae	*Curtonotus convexiusculus*+	(Marsham)	Phytophagous	0	0	0	0	2	4	6
Carabidae	*Demetrias atricapillus*	(Linnaeus)	Zoophagous	0	0	0	1	2	0	3
Carabidae	*Elaphrus riparius*+	(Linnaeus)	Zoophagous	0	0	0	1	0	0	1
Carabidae	*Harpalus affinis*	(Schrank)	Phytophagous	18	19	16	24	32	13	122
Carabidae	*Harpalus anxius*+	(Duftschmid)	Phytophagous	1	0	0	0	0	0	1
Carabidae	*Harpalus attenuatus*+	Stephens	Phytophagous	2	0	1	0	0	0	3
Carabidae	*Harpalus rubripes*+	(Duftschmid)	Phytophagous	2	1	0	3	8	2	16
Carabidae	*Harpalus rufipes*	(De Geer)	Phytophagous	1	1	6	7	16	66	97
Carabidae	*Harpalus tardus*+	(Panzer)	Phytophagous	3	3	5	3	13	30	57
Carabidae	*Leistus ferrugineus*	(Linnaeus)	Zoophagous	6	4	7	13	2	14	46
Carabidae	*Loricera pilicornis*	(Fabricius)	Zoophagous	1	0	1	0	0	1	3
Carabidae	*Microlestes maurus*	(Sturm)	Zoophagous	7	21	7	12	52	14	113
Carabidae	*Microlestes minutulus*+	(Goeze)	Zoophagous	0	1	0	0	1	0	2
Carabidae	*Nebria brevicollis*	(Fabricius)	Zoophagous	0	5	7	2	5	5	24
Carabidae	*Notiophilus biguttatus*	(Fabricius)	Zoophagous	4	0	0	0	1	1	6
Carabidae	*Notiophilus palustris*	(Duftschmid)	Zoophagous	1	1	0	13	6	8	29
Carabidae	*Notiophilus substriatus*+	Waterhouse, G.R.	Zoophagous	1	1	1	3	2	6	14
Carabidae	*Ophonus ardosiacus*+	(Lutschnik)	Phytophagous	10	23	6	11	12	8	70
Carabidae	*Ophonus azureus*+	(Fabricius)	Phytophagous	3	4	2	1	1	1	12
Carabidae	*Ophonus puncticeps*	Stephens	Phytophagous	0	0	0	2	0	1	3

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Carabidae	*Ophonus rufibarbis*	(Fabricius)	Phytophagous	44	22	17	54	46	19	202
Carabidae	*Oxypselaphus obscurus*	(Herbst)	Zoophagous	0	1	0	0	1	1	3
Carabidae	*Panagaeus bipustulatus*[+]	(Fabricius)	Zoophagous	2	1	2	7	0	0	12
Carabidae	*Paradromius linearis*[+]	(Olivier)	Zoophagous	2	0	1	1	6	6	16
Carabidae	*Philorhizus melanocephalus*[+]	(Dejean)	Zoophagous	0	0	0	1	0	2	3
Carabidae	*Platyderus depressus*[+]	(Audinet-Serville)	Zoophagous	0	0	0	0	2	1	3
Carabidae	*Poecilus cupreus*[+]	(Linnaeus)	Zoophagous	7	21	5	4	8	10	55
Carabidae	*Pterostichus macer*	(Marsham)	Zoophagous	4	6	1	6	14	5	36
Carabidae	*Pterostichus madidus*	(Fabricius)	Zoophagous	26	20	7	179	218	130	580
Carabidae	*Pterostichus nigrita*	(Paykull)	Zoophagous	1	0	0	0	0	0	1
Carabidae	*Pterostichus strenuus*	(Panzer)	Zoophagous	3	0	0	2	0	2	7
Carabidae	*Scybalicus oblongiusculus*[+]	(Dejean)	Phytophagous	1	2	2	3	18	15	41
Carabidae	*Stenolophus mixtus*	(Herbst)	Phytophagous	0	1	0	2	1	2	6
Carabidae	*Stomis pumicatus*	(Panzer)	Zoophagous	16	4	6	3	4	5	38
Carabidae	*Syntomus obscuroguttatus*	(Duftschmid)	Zoophagous	0	2	0	0	0	0	2
Carabidae	*Trechus quadristriatus*	(Schrank)	Zoophagous	3	1	3	3	2	3	15
Cerambycidae	*Gracilia minuta*	(Fabricius)	Phytophagous	0	0	0	0	1	0	1
Cerambycidae	*Phytoecia cylindrica*	(Linnaeus)	Phytophagous	0	0	0	1	0	0	1
Cerambycidae	*Pseudovadonia livida*	(Fabricius)	Phytophagous	0	0	0	0	0	1	1
Chrysomelidae	*Altica carinthiaca*	Weise	Phytophagous	0	0	0	4	0	0	4
Chrysomelidae	*Altica palustris*	Weise	Phytophagous	0	0	0	0	0	1	1
Chrysomelidae	*Aphthona euphorbiae*	(Schrank)	Phytophagous	1	1	1	0	0	1	4
Chrysomelidae	*Aphthona nigriceps*	(Redtenbacher)	Phytophagous	0	0	0	1	2	4	7
Chrysomelidae	*Batophila aerata*	(Marsham)	Phytophagous	1	0	1	1	1	3	7
Chrysomelidae	*Batophila rubi*	(Paykull)	Phytophagous	0	0	0	0	1	0	1
Chrysomelidae	*Bruchidius imbricornis*	(Panzer)	Phytophagous	0	1	0	0	2	0	3
Chrysomelidae	*Bruchus loti*	Paykull	Phytophagous	0	0	0	2	0	1	3

				2015			2016			
Family	Taxon	Authority	Trophic group	Control	RBF	Thanet	Control	RBF	Thanet	Total
Chrysomelidae	*Cassida rubiginosa*	Müller, O.F.	Phytophagous	0	0	0	0	4	5	9
Chrysomelidae	*Cassida vibex*	Linnaeus	Phytophagous	0	1	0	1	0	2	4
Chrysomelidae	*Cassida viridis*	Linnaeus	Phytophagous	0	1	0	0	0	0	1
Chrysomelidae	*Chaetocnema concinna*	(Marsham)	Phytophagous	2	10	1	4	3	9	29
Chrysomelidae	*Chaetocnema hortensis*	(Fourcroy)	Phytophagous	0	0	0	1	0	0	1
Chrysomelidae	*Chaetocnema picipes*	Stephens	Phytophagous	0	0	1	0	0	0	1
Chrysomelidae	*Chrysolina banksii*	(Fabricius)	Phytophagous	0	0	0	1	0	0	1
Chrysomelidae	*Chrysolina oricalcia*	(Müller)	Phytophagous	0	0	0	0	2	0	2
Chrysomelidae	*Lema cyanella*	(Linnaeus)	Phytophagous	0	0	0	0	0	1	1
Chrysomelidae	*Longitarsus dorsalis*	(Fabricius)	Phytophagous	0	0	0	2	222	17	241
Chrysomelidae	*Longitarsus gracilis*	Kutschera	Phytophagous	0	0	0	0	1	0	1
Chrysomelidae	*Longitarsus jacobaeae*	(Waterhouse, G.R.)	Phytophagous	0	0	0	0	4	2	6
Chrysomelidae	*Longitarsus luridus*	(Scopoli)	Phytophagous	0	0	0	0	1	1	2
Chrysomelidae	*Longitarsus melanocephalus*	(De Geer)	Phytophagous	0	0	0	1	0	0	1
Chrysomelidae	*Longitarsus parvulus*	(Paykull)	Phytophagous	0	0	1	0	0	0	1
Chrysomelidae	*Longitarsus pellucidus*	(Foudras)	Phytophagous	0	0	0	0	1	0	1
Chrysomelidae	*Longitarsus pratensis*	(Panzer)	Phytophagous	2	0	2	11	0	0	15
Chrysomelidae	*Longitarsus rubiginosus*	(Foudras)	Phytophagous	1	6	7	0	4	1	19
Chrysomelidae	*Longitarsus suturellus*	(Duftschmid)	Phytophagous	7	4	9	10	48	19	97
Chrysomelidae	*Neocrepidodera ferruginea*	(Scopoli)	Phytophagous	1	0	2	20	28	17	68
Chrysomelidae	*Neocrepidodera transversa*	(Marsham)	Phytophagous	0	0	1	0	0	0	1
Chrysomelidae	*Phaedon tumidulus*	(Germar)	Phytophagous	2	0	0	23	38	51	114
Chrysomelidae	*Phyllotreta atra*	(Fabricius)	Phytophagous	0	8	0	0	183	66	257
Chrysomelidae	*Phyllotreta consobrina*	(Curtis)	Phytophagous	0	33	4	0	60	3	100
Chrysomelidae	*Phyllotreta cruciferae*	(Goeze)	Phytophagous	0	24	8	0	4	0	36
Chrysomelidae	*Phyllotreta diademata*	Foudras	Phytophagous	0	44	0	0	39	4	87
Chrysomelidae	*Phyllotreta nemorum*	(Linnaeus)	Phytophagous	0	0	0	0	0	3	3

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Chrysomelidae	*Phyllotreta nigripes*	(Fabricius)	Phytophagous	0	15	0	0	67	13	95
Chrysomelidae	*Phyllotreta undulata*	Kutschera	Phytophagous	0	82	0	0	14	7	103
Chrysomelidae	*Phyllotreta vittula*	(Redtenbacher)	Phytophagous	0	0	0	0	1	1	2
Chrysomelidae	*Podagrica fuscipes*	(Fabricius)	Phytophagous	2	3	0	1	13	17	36
Chrysomelidae	*Psylliodes affinis*	(Paykull)	Phytophagous	0	0	0	0	1	0	1
Chrysomelidae	*Psylliodes chrysocephala*	(Linnaeus)	Phytophagous	2	12	2	2	30	1	49
Chrysomelidae	*Sphaeroderma testaceum*	(Fabricius)	Phytophagous	2	3	7	1	19	14	46
Coccinellidae	*Coccinella septempunctata*	Linnaeus	Predatory	0	4	5	0	2	1	12
Coccinellidae	*Hippodamia variegata*	(Goeze)	Predatory	0	0	1	0	0	0	1
Coccinellidae	*Nephus redtenbacheri*	(Mulsant)	Predatory	0	0	0	0	0	1	1
Coccinellidae	*Psyllobora vigintiduopunctata*	(Linnaeus)	Predatory	0	4	2	0	0	6	12
Coccinellidae	*Rhyzobius litura*	(Fabricius)	Predatory	0	0	2	3	13	9	27
Coccinellidae	*Scymnus haemorrhoidalis*	Herbst	Predatory	0	0	0	0	1	0	1
Coccinellidae	*Subcoccinella vigintiquattuorpunctata*	(Linnaeus)	Predatory	15	6	5	15	19	22	82
Coccinellidae	*Tytthaspis sedecimpunctata*	(Linnaeus)	Predatory	9	5	6	46	46	65	177
Corylophidae	*Corylophus sublaevipennis*	Jacquelin du Val	Detritivorous	0	1	0	0	0	0	1
Corylophidae	*Orthoperus brunnipes*	(Gyllenhal)	Detritivorous	0	0	0	1	7	4	12
Corylophidae	*Sericoderus brevicornis*	Matthews, A.	Detritivorous	0	0	0	2	1	2	5
Corylophidae	*Sericoderus lateralis*	(Gyllenhal)	Detritivorous	0	0	0	0	1	0	1
Cryptophagidae	*Atomaria rubella*	Heer	Detritivorous	1	1	0	0	5	2	9
Cryptophagidae	*Atomaria testacea*	Stephens	Detritivorous	1	1	0	0	1	0	3
Cryptophagidae	*Cryptophagus distinguendus*	Sturm	Detritivorous	0	0	0	0	1	0	1
Cryptophagidae	*Cryptophagus puncticollis*	Lucas	Detritivorous	0	0	0	0	0	1	1
Cryptophagidae	*Ootypus globosus*	(Waltl)	Detritivorous	5	2	1	1	5	0	14
Curculionidae	*Anthonomus rubi*	(Herbst)	Phytophagous	0	0	0	0	0	1	1
Curculionidae	*Barypeithes pellucidus*	(Boheman)	Phytophagous	22	21	2	2	11	4	62
Curculionidae	*Ceutorhynchus contractus*	(Marsham)	Phytophagous	0	0	0	0	11	5	16

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Curculionidae	*Ceutorhynchus erysimi*	(Fabricius)	Phytophagous	0	0	0	0	3	0	3
Curculionidae	*Ceutorhynchus obstrictus*	(Marsham)	Phytophagous	0	4	0	0	5	2	11
Curculionidae	*Ceutorhynchus pallidactylus*	(Marsham)	Phytophagous	5	131	10	0	19	0	165
Curculionidae	*Ceutorhynchus typhae*	(Herbst)	Phytophagous	0	1	0	0	0	0	1
Curculionidae	*Euophryum confine*	(Broun)	Phytophagous	1	0	0	0	0	0	1
Curculionidae	*Gymnetron pascuorum*	(Germar)	Phytophagous	1	0	0	1	1	3	6
Curculionidae	*Hadroplontus litura*	(Fabricius)	Phytophagous	2	0	0	0	4	6	12
Curculionidae	*Hypera fuscocinerea*⁺	(Marsham)	Phytophagous	0	0	0	0	2	0	2
Curculionidae	*Hypera meles*	(Fabricius)	Phytophagous	0	0	0	1	0	0	1
Curculionidae	*Hypera nigrirostris*	(Fabricius)	Phytophagous	0	0	0	0	0	1	1
Curculionidae	*Hypera plantaginis*	(De Geer)	Phytophagous	0	0	1	0	0	1	2
Curculionidae	*Hypera postica*	(Gyllenhal)	Phytophagous	0	2	0	2	0	1	5
Curculionidae	*Larinus planus*	(Fabricius)	Phytophagous	0	0	0	0	1	0	1
Curculionidae	*Leiosoma deflexum*	(Panzer)	Phytophagous	0	1	0	1	1	0	3
Curculionidae	*Mecinus pyraster*	(Herbst)	Phytophagous	0	1	0	0	0	0	1
Curculionidae	*Nedyus quadrimaculatus*	(Linnaeus)	Phytophagous	0	1	0	0	2	0	3
Curculionidae	*Neliocarus faber*	(Herbst)	Phytophagous	0	0	0	0	0	1	1
Curculionidae	*Neophytobius quadrinodosus*	(Gyllenhal)	Phytophagous	1	0	0	2	0	1	4
Curculionidae	*Orthochaetes setiger*	(Beck)	Phytophagous	0	0	0	1	0	0	1
Curculionidae	*Otiorhynchus ligneus*⁺	(Olivier)	Phytophagous	0	0	0	0	2	1	3
Curculionidae	*Otiorhynchus ovatus*⁺	(Linnaeus)	Phytophagous	0	0	1	0	0	0	1
Curculionidae	*Otiorhynchus raucus*⁺	(Fabricius)	Phytophagous	0	1	0	0	0	3	4
Curculionidae	*Otiorhynchus rugosostriatus*	(Goeze)	Phytophagous	0	0	2	2	29	2	35
Curculionidae	*Otiorhynchus sulcatus*	(Fabricius)	Phytophagous	0	0	0	0	0	1	1
Curculionidae	*Parethelcus pollinarius*	(Forster)	Phytophagous	0	0	0	0	1	1	2
Curculionidae	*Pelenomus quadrituberculatus*	(Fabricius)	Phytophagous	1	0	0	1	0	0	2
Curculionidae	*Phyllobius pyri*	(Linnaeus)	Phytophagous	0	0	0	0	1	0	1

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Curculionidae	*Phyllobius viridicollis*	(Fabricius)	Phytophagous	0	0	0	1	0	1	2
Curculionidae	*Rhinocyllus conicus*	(Frölich)	Phytophagous	0	0	0	0	1	2	3
Curculionidae	*Rhinoncus pericarpius*	(Linnaeus)	Phytophagous	0	0	0	0	0	1	1
Curculionidae	*Romualdius angustisetulus*[+]	(Hansen)	Phytophagous	0	0	0	0	1	0	1
Curculionidae	*Sitona cylindricollis*	(Fåhraeus)	Phytophagous	0	0	1	0	0	0	1
Curculionidae	*Sitona hispidulus*	(Fabricius)	Phytophagous	0	0	0	2	0	0	2
Curculionidae	*Sitona humeralis*	Stephens	Phytophagous	0	1	2	0	0	0	3
Curculionidae	*Sitona lepidus*	Gyllenhal	Phytophagous	0	0	0	1	0	0	1
Curculionidae	*Sitona lineatus*	(Linnaeus)	Phytophagous	3	1	3	7	4	11	29
Curculionidae	*Trichosirocalus troglodytes*	(Fabricius)	Phytophagous	0	1	0	0	2	0	3
Curculionidae	*Tychius picirostris*	(Fabricius)	Phytophagous	0	0	0	0	0	1	1
Curculionidae	*Tychius tibialis*	Boheman	Phytophagous	0	0	0	1	0	0	1
Curculionidae	*Zacladus exiguus*	(Olivier)	Phytophagous	1	0	0	0	1	2	4
Dermestidae	*Megatoma undata*	(Linnaeus)	Detritivorous	0	1	0	0	1	0	2
Dytiscidae	*Colymbetes fuscus*	(Linnaeus)	Predatory	1	0	0	0	0	0	1
Dytiscidae	*Hydaticus seminiger*	(De Geer)	Predatory	0	0	0	1	0	0	1
Dytiscidae	*Rhantus frontalis*	(Marsham)	Predatory	0	0	1	0	0	0	1
Elateridae	*Agriotes lineatus*	(Linnaeus)	Phytophagous	3	4	7	0	2	5	21
Elateridae	*Agriotes sputator*	(Linnaeus)	Phytophagous	76	30	33	130	84	86	439
Elateridae	*Agrypnus murinus*	(Linnaeus)	Phytophagous	1	0	0	2	1	2	6
Elateridae	*Athous campyloides*	Newman	Phytophagous	0	1	0	1	0	0	2
Helophoridae	*Helophorus brevipalpis*	Bedel	Detritivorous	1	0	0	0	0	0	1
Helophoridae	*Helophorus rufipes*[+]	(Bosc d'Antic)	Detritivorous	0	1	0	0	0	0	1
Heteroceridae	*Heterocerus fenestratus*	(Thunberg)	Phytophagous	0	0	0	0	0	1	1
Histeridae	*Hister unicolor*	Linnaeus	Predatory	0	0	0	0	0	1	1
Histeridae	*Kissister minimus*	(Aubé)	Predatory	2	0	1	3	0	0	6
Histeridae	*Margarinotus brunneus*	(Fabricius)	Predatory	0	0	0	1	0	0	1

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Histeridae	*Margarinotus purpurascens*	(Herbst)	Predatory	2	0	0	2	2	0	6
Hydrophilidae	*Cercyon melanocephalus*	(Linnaeus)	Detritivorous	0	0	0	1	0	0	1
Hydrophilidae	*Cercyon pygmaeus*	(Illiger)	Detritivorous	0	0	0	1	0	0	1
Hydrophilidae	*Cercyon tristis*	(Illiger)	Detritivorous	0	2	1	0	0	0	3
Hydrophilidae	*Megasternum concinnum*	(Marsham)	Detritivorous	26	2	4	16	8	4	60
Hydrophilidae	*Sphaeridium lunatum*	Fabricius	Detritivorous	0	1	0	0	0	0	1
Hydrophilidae	*Sphaeridium marginatum*	Fabricius	Detritivorous	0	0	0	1	0	0	1
Kateretidae	*Brachypterus glaber*	(Newman)	Phytophagous	1	3	0	0	1	0	5
Kateretidae	*Brachypterus urticae*	(Fabricius)	Phytophagous	0	2	0	0	0	0	2
Latridiidae	*Cartodere bifasciata*	(Reitter)	Detritivorous	0	0	0	1	2	1	4
Latridiidae	*Corticaria impressa*	(Olivier)	Detritivorous	8	5	4	11	9	9	46
Latridiidae	*Corticarina minuta*	(Fabricius)	Detritivorous	7	3	8	0	0	0	18
Latridiidae	*Cortinicara gibbosa*	(Herbst)	Detritivorous	1	2	1	0	1	0	5
Latridiidae	*Enicmus transversus*	(Olivier)	Detritivorous	19	22	26	4	38	24	133
Latridiidae	*Melanophthalma suturalis*[+]	(Mannerheim)	Detritivorous	0	0	0	0	2	4	6
Leiodidae	*Agathidium laevigatum*	Erichson	Detritivorous	3	1	0	12	8	6	30
Leiodidae	*Catops fuscus*	(Panzer)	Detritivorous	0	0	0	1	0	0	1
Leiodidae	*Catops grandicollis*	Erichson	Detritivorous	0	0	0	1	0	0	1
Leiodidae	*Catops morio*	(Fabricius)	Detritivorous	5	1	2	1	1	1	11
Leiodidae	*Choleva agilis*	(Illiger)	Detritivorous	1	0	0	1	2	1	5
Leiodidae	*Choleva oblonga*	Latreille	Detritivorous	2	1	0	0	0	0	3
Leiodidae	*Nargus anisotomoides*	(Spence)	Detritivorous	2	0	0	0	0	1	3
Leiodidae	*Nargus velox*	(Spence)	Detritivorous	0	1	0	8	3	8	20
Leiodidae	*Ptomaphagus medius*	Rey	Detritivorous	0	0	0	2	2	1	5
Leiodidae	*Ptomaphagus subvillosus*	(Goeze)	Detritivorous	28	17	4	22	12	14	97
Leiodidae	*Ptomaphagus varicornis*	(Rosenhauer)	Detritivorous	0	0	0	0	1	2	3
Leiodidae	*Sciodrepoides watsoni*	(Spence)	Detritivorous	0	0	4	0	2	1	7

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Malachiidae	*Cordylepherus viridis*	(Fabricius)	Predatory	0	0	1	0	0	0	1
Mordellidae	*Mordellistena parvula*	(Gyllenhal)	Phytophagous	0	0	1	0	4	0	5
Nitidulidae	*Glischrochilus hortensis*	(Geoffroy in Fourcroy)	Phytophagous	0	0	0	0	1	0	1
Nitidulidae	*Meligethes aeneus*	(Fabricius)	Phytophagous	1	9	2	0	40	5	57
Nitidulidae	*Meligethes fulvipes*	Brisout de Barneville	Phytophagous	0	0	0	0	1	0	1
Nitidulidae	*Meligethes rotundicollis*	Brisout de Barneville	Phytophagous	0	12	0	0	1	1	14
Nitidulidae	*Soronia grisea*	(Linnaeus)	Phytophagous	0	0	0	0	4	0	4
Nitidulidae	*Thalycra fervida*	(Olivier)	Phytophagous	0	0	0	0	1	0	1
Oedemeridae	*Oedemera nobilis*	(Scopoli)	Phytophagous	1	0	0	0	4	0	5
Phalacridae	*Olibrus aeneus*	(Fabricius)	Phytophagous	0	0	0	0	1	0	1
Phalacridae	*Phalacrus championi*	Guillebeau	Phytophagous	0	0	0	1	1	1	3
Phalacridae	*Phalacrus fimetarius*	(Fabricius)	Phytophagous	0	0	0	0	0	1	1
Phalacridae	*Stilbus oblongus*	(Erichson)	Phytophagous	0	0	0	0	1	0	1
Ptiliidae	*Acrotrichis fascicularis*	(Herbst)	Detritivorous	0	0	1	0	0	0	1
Ptiliidae	*Acrotrichis sitkaensis*	(Motschulsky)	Detritivorous	0	1	0	0	0	0	1
Ptiliidae	*Ptenidium fuscicorne*	Erichson	Detritivorous	0	0	1	0	0	0	1
Ptiliidae	*Acrotrichis fascicularis*	(Herbst)	Detritivorous	1	0	0	1	2	3	7
Ptiliidae	*Acrotrichis sitkaensis*	(Motschulsky)	Detritivorous	3	0	0	0	0	0	3
Ptiliidae	*Ptenidium intermedium*	Wankowicz	Detritivorous	0	0	0	0	0	2	2
Ptiliidae	*Ptenidium pusillum*	(Gyllenhal)	Detritivorous	0	1	0	0	0	1	2
Scarabaeidae	*Aphodius erraticus*	(Linnaeus)	Detritivorous	2	0	1	0	0	0	3
Scarabaeidae	*Aphodius foetidus*	(Herbst)	Detritivorous	0	0	0	1	0	0	1
Scarabaeidae	*Aphodius plagiatus*	(Linnaeus)	Detritivorous	2	1	0	6	0	3	12
Scarabaeidae	*Aphodius prodromus*	(Brahm)	Detritivorous	0	0	0	1	0	0	1
Scirtidae	*Cyphon coarctatus*	Paykull	Phytophagous	0	1	0	0	0	0	1
Scirtidae	*Cyphon laevipennis*	Tournier	Phytophagous	0	0	0	0	0	1	1
Silphidae	*Nicrophorus interruptus*	Stephens	Detritivorous	0	0	2	0	0	0	2

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Silphidae	*Nicrophorus vespillo*	(Linnaeus)	Detritivorous	3	0	1	3	1	1	9
Silphidae	*Silpha atrata*	Linnaeus	Predatory	0	1	0	3	12	7	23
Silphidae	*Silpha laevigata*	Fabricius	Predatory	0	0	0	1	3	2	6
Silphidae	*Silpha tristis*	Illiger	Predatory	3	0	0	3	6	0	12
Silphidae	*Thanatophilus sinuatus*	(Fabricius)	Detritivorous	0	1	4	0	0	0	5
Silvanidae	*Psammoecus bipunctatus*	(Fabricius)	Detritivorous	0	0	0	0	0	1	1
Staphylinidae	*Acrolocha sulcula*	(Stephens)	Predatory	0	3	0	0	0	0	3
Staphylinidae	*Acrotona muscorum*	(Brisout de Barneville)	Predatory	1	0	0	0	0	0	1
Staphylinidae	*Acrotona parens*	(Mulsant & Rey)	Predatory	0	0	1	0	0	0	1
Staphylinidae	*Aleochara bipustulata*	(Linnaeus)	Predatory	1	62	58	0	4	5	130
Staphylinidae	*Aleochara brevipennis*	Gravenhorst	Predatory	1	0	0	1	2	1	5
Staphylinidae	*Aleochara curtula*	(Goeze)	Predatory	0	0	3	0	1	3	7
Staphylinidae	*Aleochara lanuginosa*	Gravenhorst	Predatory	1	0	0	0	0	0	1
Staphylinidae	*Aleochara sparsa*	Heer	Predatory	0	1	0	0	0	0	1
Staphylinidae	*Aloconota gregaria*	(Erichson)	Predatory	5	5	3	4	6	3	26
Staphylinidae	*Amarochara forticornis*	(Lacordaire)	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Amischa analis*	(Gravenhorst)	Predatory	9	6	4	9	7	5	40
Staphylinidae	*Amischa decipiens*	(Sharp)	Predatory	3	3	1	3	6	0	16
Staphylinidae	*Anotylus insecatus*	(Gravenhorst)	Detritivorous	0	9	4	0	23	10	46
Staphylinidae	*Anotylus inustus*[+]	(Gravenhorst)	Detritivorous	0	1	3	0	0	0	4
Staphylinidae	*Anotylus rugosus*	(Fabricius)	Detritivorous	5	7	6	1	0	2	21
Staphylinidae	*Anotylus sculpturatus*	(Gravenhorst)	Detritivorous	2	2	0	0	0	1	5
Staphylinidae	*Anotylus tetracarinatus*	(Block)	Detritivorous	0	1	1	0	0	0	2
Staphylinidae	*Astenus immaculatus*	Stephens	Predatory	1	4	2	2	4	4	17
Staphylinidae	*Astenus lyonessius*[+]	(Joy)	Predatory	0	2	0	0	0	1	3
Staphylinidae	*Atheta aquatica*	(Thomson)	Predatory	0	0	0	1	1	0	2
Staphylinidae	*Atheta crassicornis*	(Fabricius)	Predatory	0	0	0	0	1	0	1

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Staphylinidae	*Atheta triangulum*	(Kraatz)	Predatory	0	1	0	0	0	0	1
Staphylinidae	*Atheta xanthopus*	(Thomson)	Predatory	0	1	0	0	2	4	7
Staphylinidae	*Bolitobius castaneus*	(Stephens)	Predatory	2	2	2	5	6	2	19
Staphylinidae	*Brachida exigua*	(Heer)	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Brachygluta fossulata*	(Reichenbach)	Predatory	2	5	2	6	6	9	30
Staphylinidae	*Brachygluta haematica*	(Reichenbach)	Predatory	0	0	1	0	0	0	1
Staphylinidae	*Brachygluta waterhousei*	(Rye)	Predatory	0	0	1	0	0	0	1
Staphylinidae	*Callicerus obscurus*	Gravenhorst	Predatory	2	0	0	0	2	2	6
Staphylinidae	*Callicerus rigidicornis*	(Erichson)	Predatory	1	2	0	8	9	14	34
Staphylinidae	*Calodera protensa*	Mannerheim	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Carpelimus corticinus*[+]	(Gravenhorst)	Detritivorous	0	0	0	3	0	0	3
Staphylinidae	*Carpelimus incongruus*[+]	Steel	Detritivorous	0	0	0	0	1	0	1
Staphylinidae	*Carpelimus rivularis*[+]	(Motschulsky)	Detritivorous	0	1	0	0	0	0	1
Staphylinidae	*Chaetida longicornis*	(Gravenhorst)	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Cypha longicornis*	(Paykull)	Predatory	3	1	3	8	15	12	42
Staphylinidae	*Dinaraea angustula*	(Gyllenhal)	Predatory	11	20	11	5	30	22	99
Staphylinidae	*Drusilla canaliculata*	(Fabricius)	Predatory	546	51	21	763	142	150	1673
Staphylinidae	*Falagrioma thoracica*	(Stephens)	Predatory	3	2	0	5	10	5	25
Staphylinidae	*Gabrius breviventer*	(Sperk)	Predatory	1	0	0	4	2	0	7
Staphylinidae	*Gyrohypnus fracticornis*	(Müller, O.F.)	Predatory	0	1	0	0	0	0	1
Staphylinidae	*Ilyobates propinquus*	(Aubé)	Predatory	1	0	1	1	3	0	6
Staphylinidae	*Ischnosoma splendidum*	(Gravenhorst)	Predatory	2	0	0	1	0	0	3
Staphylinidae	*Lathrobium fulvipenne*	(Gravenhorst)	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Lathrobium geminum*	Kraatz	Predatory	1	0	0	1	0	0	2
Staphylinidae	*Lesteva longoelytrata*	(Goeze)	Detritivorous	0	0	0	0	0	1	1
Staphylinidae	*Lesteva sicula*	Erichson	Detritivorous	1	0	0	0	0	1	2
Staphylinidae	*Liogluta microptera*	Thomson, C.G.	Predatory	0	0	0	1	0	8	9

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Staphylinidae	*Liogluta pagana*	(Erichson)	Predatory	19	10	10	5	3	1	48
Staphylinidae	*Lobrathium multipunctum*[+]	(Gravenhorst)	Predatory	0	0	0	0	1	1	2
Staphylinidae	*Megalinus glabratus*	(Gravenhorst)	Predatory	0	0	0	0	1	0	1
Staphylinidae	*Metopsia clypeata*	(Müller, P.W.J.)	Predatory	4	0	2	2	0	1	9
Staphylinidae	*Micropeplus staphylinoides*	(Marsham)	Predatory	10	4	2	13	0	1	30
Staphylinidae	*Mocyta fungi*	(Gravenhorst)	Predatory	0	0	1	0	0	0	1
Staphylinidae	*Mycetoporus nigricollis*	Stephens	Predatory	0	2	1	2	0	1	6
Staphylinidae	*Neobisnius villosulus*[+]	(Stephens)	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Ocypus brunnipes*	(Fabricius)	Predatory	0	1	1	8	6	6	22
Staphylinidae	*Ocypus fuscatus*	(Gravenhorst)	Predatory	4	1	2	5	4	2	18
Staphylinidae	*Ocypus olens*	(Müller, O.F.)	Predatory	166	172	131	117	100	64	750
Staphylinidae	*Ocyusa picina*	(Aubé)	Predatory	0	0	1	0	0	0	1
Staphylinidae	*Oligota pumilio*	Kiesenwetter	Predatory	3	4	6	2	42	15	72
Staphylinidae	*Ontholestes murinus*	(Linnaeus)	Predatory	0	0	0	12	0	0	12
Staphylinidae	*Othius laeviusculus*	Stephens	Predatory	1	0	1	0	0	0	2
Staphylinidae	*Oxypoda brachyptera*[+]	(Stephens)	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Oxypoda procerula*[+]	Mannerheim	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Paederus littoralis*[+]	Gravenhorst	Predatory	0	0	1	0	2	0	3
Staphylinidae	*Pella limbata*	(Paykull)	Predatory	0	0	0	1	3	3	7
Staphylinidae	*Philhygra debilis*[+]	(Erichson)	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Philhygra elongatula*	(Gravenhorst)	Predatory	2	0	0	0	0	0	2
Staphylinidae	*Philhygra palustris*[+]	(Kiesenwetter)	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Philonthus carbonarius*	(Gravenhorst)	Predatory	7	1	2	20	6	2	38
Staphylinidae	*Philonthus cognatus*	Stephens	Predatory	19	0	2	7	1	2	31
Staphylinidae	*Philonthus cruentatus*	(Gmelin in Linnaeus)	Predatory	0	0	0	4	0	0	4
Staphylinidae	*Philonthus laminatus*	(Creutzer)	Predatory	2	0	0	1	0	1	4
Staphylinidae	*Philonthus marginatus*	(Müller, O.F.)	Predatory	0	1	0	1	0	0	2

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Staphylinidae	*Philonthus sanguinolentus*	(Gravenhorst)	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Philonthus splendens*	(Fabricius)	Predatory	0	0	0	3	0	0	3
Staphylinidae	*Philonthus succicola*	Thomson, C.G.	Predatory	0	1	0	1	3	0	5
Staphylinidae	*Philonthus tenuicornis*	Mulsant & Rey	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Philonthus varians*	(Paykull)	Predatory	0	0	0	3	0	0	3
Staphylinidae	*Platydracus stercorarius*	(Olivier)	Predatory	21	0	0	4	3	8	36
Staphylinidae	*Platystethus alutaceus*⁺	Thomson, C.G.	Predatory	1	2	0	0	2	3	8
Staphylinidae	*Platystethus nitens*⁺	(Sahlberg, C.R.)	Predatory	1	4	8	8	12	6	39
Staphylinidae	*Quedius curtipennis*	Bernhauer	Predatory	7	1	1	24	6	5	44
Staphylinidae	*Quedius fuliginosus*	(Gravenhorst)	Predatory	0	0	0	2	2	3	7
Staphylinidae	*Quedius levicollis*	(Brullé)	Predatory	30	0	3	21	5	1	60
Staphylinidae	*Quedius molochinus*	(Gravenhorst)	Predatory	8	0	0	1	0	1	10
Staphylinidae	*Quedius nemoralis*	Baudi	Predatory	2	0	0	0	0	0	2
Staphylinidae	*Quedius nigriceps*	Kraatz	Predatory	0	0	0	3	0	2	5
Staphylinidae	*Quedius nitipennis*	(Stephens)	Predatory	0	1	0	0	0	0	1
Staphylinidae	*Quedius picipes*	(Mannerheim)	Predatory	4	2	1	10	3	2	22
Staphylinidae	*Quedius semiaeneus*	(Stephens)	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Quedius semiobscurus*	(Marsham)	Predatory	41	3	3	23	5	1	76
Staphylinidae	*Rugilus erichsonii*	(Fauvel)	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Rugilus orbiculatus*⁺	(Paykull)	Predatory	8	0	1	3	0	0	12
Staphylinidae	*Rugilus rufipes*	Germar	Predatory	7	2	0	3	2	2	16
Staphylinidae	*Sepedophilus immaculatus*	(Stephens)	Predatory	23	1	0	14	4	9	51
Staphylinidae	*Sepedophilus marshami*	(Stephens)	Predatory	10	3	2	7	5	4	31
Staphylinidae	*Sepedophilus nigripennis*	(Stephens)	Predatory	6	1	0	9	2	3	21
Staphylinidae	*Staphylinus dimidiaticornis*	Gemminger	Predatory	9	0	0	12	0	5	26
Staphylinidae	*Stenichnus scutellaris*	(Müller, P.W.J. & Kunze)	Predatory	0	1	0	0	1	0	2
Staphylinidae	*Stenus brunnipes*	Stephens	Predatory	0	1	0	3	2	1	7

Family	Taxon	Authority	Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Staphylinidae	*Stenus clavicornis*	(Scopoli)	Predatory	1	4	0	6	2	3	16
Staphylinidae	*Stenus juno*	(Paykull)	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Stenus nanus*	Stephens	Predatory	0	0	0	1	0	0	1
Staphylinidae	*Stenus nitens*	Stephens	Predatory	1	0	0	0	0	0	1
Staphylinidae	*Stenus ossium*	Stephens	Predatory	1	0	0	0	1	0	2
Staphylinidae	*Stenus providus*	Erichson	Predatory	0	0	1	0	0	0	1
Staphylinidae	*Sunius propinquus*[+]	(Brisout de Barneville)	Predatory	2	1	0	1	1	2	7
Staphylinidae	*Tachinus flavolimbatus*	Pandellé	Predatory	1	0	0	2	0	0	3
Staphylinidae	*Tachinus marginellus*	(Fabricius)	Predatory	0	0	0	0	0	1	1
Staphylinidae	*Tachinus rufipes*	(Linnaeus)	Predatory	68	0	0	72	12	18	170
Staphylinidae	*Tachyporus atriceps*	Stephens	Predatory	0	0	0	1	0	1	2
Staphylinidae	*Tachyporus chrysomelinus*	(Linnaeus)	Predatory	0	2	3	2	0	1	8
Staphylinidae	*Tachyporus dispar*	(Paykull)	Predatory	15	6	6	34	2	5	68
Staphylinidae	*Tachyporus hypnorum*	(Fabricius)	Predatory	15	5	8	48	9	24	109
Staphylinidae	*Tachyporus nitidulus*	(Fabricius)	Predatory	38	67	51	51	77	137	421
Staphylinidae	*Tasgius ater*[+]	(Gravenhorst)	Predatory	0	0	1	0	0	0	1
Staphylinidae	*Tasgius globulifer*[+]	(Geoffroy)	Predatory	0	0	0	1	4	3	8
Staphylinidae	*Tasgius melanarius*	(Heer)	Predatory	1	0	0	1	0	0	2
Staphylinidae	*Tasgius morsitans*	(Rossi)	Predatory	2	1	1	13	3	4	24
Staphylinidae	*Tasgius winkleri*[+]	Bernhauer	Predatory	0	5	1	1	5	3	15
Staphylinidae	*Thinonoma atra*	(Gravenhorst)	Predatory	1	0	0	0	0	0	1
Staphylinidae	*Xantholinus elegans*[+]	(Olivier)	Predatory	2	4	0	8	2	3	19
Staphylinidae	*Xantholinus linearis*	(Olivier)	Predatory	13	3	3	19	19	15	72
Staphylinidae	*Xantholinus longiventris*	Heer	Predatory	5	5	2	5	2	0	19
Tenebrionidae	*Helops caeruleus*	(Linnaeus)	Detritivorous	0	0	0	0	0	1	1
Tenebrionidae	*Scaphidema metallicum*	(Fabricius)	Detritivorous	1	0	0	0	0	0	1
Throscidae	*Trixagus carinifrons*	(de Bonvouloir)	unknown	0	1	0	0	0	0	1

				2015			2016			
Family	Taxon	Authority	Trophic group	Control	RBF	Thanet	Control	RBF	Thanet	Total
Throscidae	*Trixagus obtusus*	(Curtis)	unknown	1	1	1	0	3	1	7

Table A9.5.2 Capture data for spiders from aggregate mounds experiment at Hadleigh Park, Pitsea Landfill, and Rainham Marshes, in 2015 and 2016. Treatments were 5-tonne mounds of recycled bulk fill (n=9) or Thanet sand (n=9), and unmanaged grassland (n=9). There were two pitfall traps per monitoring plot in each year, open for 16 days in May, July, and September. Plus symbol (+) by species name denotes status as an stenotopic indicator of early successional habitat in this study. Species were grouped by trophic guild (all spiders predatory).

				2015			2016			
Family	Taxon		Trophic group	Control	RBF	Thanet	Control	RBF	Thanet	Total
Agelenidae	*Tegenaria agrestis*+	(Walckenaer)	Predatory	0	3	0	0	0	1	4
Agelenidae	*Tegenaria silvestris*	L.Koch	Predatory	0	0	2	0	1	0	3
Clubionidae	*Cheiracanthium erraticum*	(Walckenaer)	Predatory	0	0	0	0	1	0	1
Clubionidae	*Clubiona phragmitis*	C.L.Koch	Predatory	0	0	0	1	0	0	1
Clubionidae	*Clubiona reclusa*	O.P.-Cambridge	Predatory	1	1	1	0	0	0	3
Clubionidae	*Clubiona subtilis*	L.Koch	Predatory	1	0	2	2	0	0	5
Dictynidae	*Argenna subnigra*+	(O.P.-Cambridge)	Predatory	1	0	0	0	0	1	2
Dysderidae	*Dysdera crocata*+	C.L.Koch	Predatory	3	20	2	0	9	1	35
Gnaphosidae	*Drassodes cupreus*+	(Blackwall)	Predatory	4	5	0	3	1	0	13
Gnaphosidae	*Drassyllus pusillus*+	(C.L.Koch)	Predatory	9	5	3	9	0	0	26
Gnaphosidae	*Haplodrassus signifer*+	(C.L.Koch)	Predatory	4	10	2	0	0	3	19
Gnaphosidae	*Micaria pulicaria*	(Sundevall)	Predatory	0	4	1	2	2	0	9
Gnaphosidae	*Trachyzelotes pedestris*+	(C.L.Koch)	Predatory	1	7	3	2	10	4	27
Gnaphosidae	*Zelotes apricorum*+	(L.Koch)	Predatory	3	3	3	0	4	2	15
Gnaphosidae	*Zelotes latreillei*+	(Simon)	Predatory	7	2	0	18	4	5	36
Hahniidae	*Hahnia nava*	(Blackwall)	Predatory	23	11	14	21	23	15	107
Linyphiidae	*Bathyphantes gracilis*	(Blackwall)	Predatory	2	0	1	8	1	2	14
Linyphiidae	*Bathyphantes parvulus*	(Westring)	Predatory	11	3	1	8	2	0	25
Linyphiidae	*Centromerus capucinus*+	(Simon)	Predatory	0	0	0	0	1	0	1
Linyphiidae	*Ceratinella brevipes*	(Westring)	Predatory	1	0	0	0	0	0	1
Linyphiidae	*Ceratinopsis stativa*	(Simon)	Predatory	3	0	0	3	0	0	6
Linyphiidae	*Cnephalocotes obscurus*	(Blackwall)	Predatory	1	0	0	2	0	1	4
Linyphiidae	*Dicymbium nigrum*	(Blackwall)	Predatory	0	0	0	1	0	0	1

Family	Taxon		Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Linyphiidae	*Diplocephalus graecus*	(O.P.-Cambridge)	Predatory	0	0	0	0	0	1	1
Linyphiidae	*Diplostyla concolor*	(Wider)	Predatory	1	5	5	3	7	1	22
Linyphiidae	*Erigone atra*	Blackwall	Predatory	4	6	4	7	0	2	23
Linyphiidae	*Erigone dentipalpis*	(Wider)	Predatory	0	1	2	3	1	5	12
Linyphiidae	*Gnathonarium dentatum*	(Wider)	Predatory	0	0	1	0	0	0	1
Linyphiidae	*Gongylidiellum vivum*	(O.P.-Cambridge)	Predatory	0	0	1	0	1	1	3
Linyphiidae	*Lophomma punctatum*	(Blackwall)	Predatory	0	0	0	2	1	0	3
Linyphiidae	*Maso sundevalli*	(Westring)	Predatory	2	1	0	0	0	0	3
Linyphiidae	*Meioneta beata*	(O.P.-Cambridge)	Predatory	2	2	1	2	2	2	11
Linyphiidae	*Meioneta rurestris*	(C.L.Koch)	Predatory	1	0	0	2	0	0	3
Linyphiidae	*Meioneta saxatilis*	(Blackwall)	Predatory	0	0	0	0	1	1	2
Linyphiidae	*Meioneta simplicitarsis*	(Simon)	Predatory	0	0	1	1	0	0	2
Linyphiidae	*Micrargus herbigradus*	(Blackwall)	Predatory	0	2	0	0	1	1	4
Linyphiidae	*Micrargus subaequalis*	(Westring)	Predatory	1	6	4	1	0	1	13
Linyphiidae	*Microlinyphia pusilla*	(Sundevall)	Predatory	0	0	2	0	1	0	3
Linyphiidae	*Monocephalus fuscipes*	(Blackwall)	Predatory	0	0	0	0	0	1	1
Linyphiidae	*Neriene clathrata*	(Sundevall)	Predatory	0	3	0	1	1	0	5
Linyphiidae	*Oedothorax apicatus*	(Blackwall)	Predatory	1	0	0	2	0	0	3
Linyphiidae	*Oedothorax fuscus*	(Blackwall)	Predatory	0	0	0	4	2	1	7
Linyphiidae	*Oedothorax retusus*	(Westring)	Predatory	1	0	3	2	2	1	9
Linyphiidae	*Ostearius melanopygius*	(O.P.-Cambridge)	Predatory	0	1	1	0	0	0	2
Linyphiidae	*Palliduphantes ericaeus*	(Blackwall)	Predatory	1	0	1	2	0	0	4
Linyphiidae	*Palliduphantes pallidus*	(O.P.-Cambridge)	Predatory	0	0	0	0	1	0	1
Linyphiidae	*Panamomops sulcifrons*	(Wider)	Predatory	7	5	2	10	3	3	30
Linyphiidae	*Pelecopsis parallela*	(Wider)	Predatory	4	0	1	6	0	0	11
Linyphiidae	*Pocadicnemis juncea*	Locket & Millidge	Predatory	1	0	2	1	0	1	5
Linyphiidae	*Porrhomma pygmaeum*	(Blackwall)	Predatory	0	0	1	1	0	0	2

Family	Taxon		Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Linyphiidae	*Stemonyphantes lineatus*	(Linnaeus)	Predatory	1	0	0	0	0	0	1
Linyphiidae	*Syedra gracilis*	(Menge)	Predatory	3	1	0	3	0	2	9
Linyphiidae	*Tenuiphantes tenebricola*	(Wider)	Predatory	1	0	0	0	0	0	1
Linyphiidae	*Tenuiphantes tenuis*	(Blackwall)	Predatory	13	5	7	27	28	29	109
Linyphiidae	*Tenuiphantes zimmermanni*	Bertkau	Predatory	0	0	1	0	0	0	1
Linyphiidae	*Tiso vagans*	(Blackwall)	Predatory	0	0	0	0	1	0	1
Linyphiidae	*Troxochrus scabriculus*⁺	(Westring)	Predatory	0	0	1	0	0	0	1
Linyphiidae	*Walckenaeria acuminata*	Blackwall	Predatory	0	0	0	1	0	0	1
Linyphiidae	*Walckenaeria antica*	(Wider)	Predatory	1	1	1	2	1	1	7
Linyphiidae	*Walckenaeria atrotibialis*	(O.P.-Cambridge)	Predatory	4	3	1	0	0	0	8
Linyphiidae	*Walckenaeria capito*⁺	(Westring)	Predatory	0	1	0	0	0	0	1
Linyphiidae	*Walckenaeria dysderoides*	(Wider)	Predatory	0	3	1	0	1	0	5
Linyphiidae	*Walckenaeria nudipalpis*	(Westring)	Predatory	0	0	0	0	1	0	1
Linyphiidae	*Walckenaeria unicornis*	O.P.-Cambridge	Predatory	1	1	0	0	0	0	2
Liocranidae	*Agraecina striata*	(Kulczynski)	Predatory	6	3	2	1	0	0	12
Liocranidae	*Agroeca inopina*	O.P.-Cambridge	Predatory	2	7	4	0	0	0	13
Liocranidae	*Agroeca proxima*	(O.P.-Cambridge	Predatory	0	0	0	1	0	0	1
Liocranidae	*Phrurolithus festivus*	(C.L.Koch)	Predatory	1	28	5	2	13	2	51
Liocranidae	*Phrurolithus minimus*⁺	C.L.Koch	Predatory	0	1	2	0	3	1	7
Lycosidae	*Alopecosa pulverulenta*	(Clerck)	Predatory	196	40	19	399	91	136	881
Lycosidae	*Pardosa agrestis*	(Westring)	Predatory	3	1	1	0	0	0	5
Lycosidae	*Pardosa nigriceps*	(Thorell)	Predatory	24	51	49	43	45	72	284
Lycosidae	*Pardosa palustris*	(Linnaeus)	Predatory	116	22	18	14	5	8	183
Lycosidae	*Pardosa prativaga*	(L.Koch)	Predatory	178	89	37	120	59	51	534
Lycosidae	*Pardosa pullata*	(Clerck)	Predatory	87	27	4	114	17	37	286
Lycosidae	*Pirata uliginosus*	(Thorell)	Predatory	1	0	0	0	0	0	1
Lycosidae	*Trochosa ruricola*	(De Geer)	Predatory	11	16	11	21	16	15	90

Family	Taxon		Trophic group	2015			2016			Total
				Control	RBF	Thanet	Control	RBF	Thanet	
Lycosidae	*Trochosa terricola*	Thorell	Predatory	34	16	4	84	26	30	194
Mimetidae	*Ero furcata*	(Villers)	Predatory	0	1	1	0	0	0	2
Philodromidae	*Philodromus cespitum*	(Walckenaer)	Predatory	0	0	0	0	0	2	2
Philodromidae	*Thanatus striatus*	C.L.Koch	Predatory	0	0	0	0	0	1	1
Philodromidae	*Tibellus oblongus*	(Walckenaer)	Predatory	0	1	0	0	0	1	2
Pisauridae	*Pisaura mirabilis*	(Clerck)	Predatory	0	0	1	0	0	0	1
Salticidae	*Euophrys frontalis*	(Walckenaer)	Predatory	0	1	2	0	0	0	3
Salticidae	*Sibianor aurocinctus*⁺	(Ohlert)	Predatory	0	0	0	1	0	1	2
Tetragnathidae	*Pachygnatha clercki*	Sundevall	Predatory	1	0	0	0	0	0	1
Tetragnathidae	*Pachygnatha degeeri*	Sundevall	Predatory	69	8	5	88	6	21	197
Theridiidae	*Crustulina guttata*⁺	(Wider)	Predatory	0	1	0	0	0	1	2
Theridiidae	*Enoplognatha latimana*	Hippa & Oksala	Predatory	1	0	0	0	0	0	1
Theridiidae	*Enoplognatha thoracica*	(Hahn)	Predatory	0	1	1	0	0	0	2
Theridiidae	*Robertus arundineti*	(O.P.-Cambridge)	Predatory	0	0	2	0	0	0	2
Theridiidae	*Robertus lividus*	(Blackwall)	Predatory	1	4	3	1	1	6	16
Theridiidae	*Theridion varians*	Hahn	Predatory	0	0	0	1	0	0	1
Thomisidae	*Ozyptila brevipes*	(Hahn)	Predatory	0	2	1	0	0	0	3
Thomisidae	*Ozyptila praticola*	(C.L.Koch)	Predatory	0	0	0	0	1	0	1
Thomisidae	*Ozyptila sanctuaria*	(O.P.-Cambridge)	Predatory	0	4	6	0	0	0	10
Thomisidae	*Ozyptila simplex*⁺	(O.P.-Cambridge)	Predatory	6	2	1	2	1	0	12
Thomisidae	*Ozyptila trux*	(Blackwall)	Predatory	0	0	0	1	0	0	1
Thomisidae	*Xysticus cristatus*	(Clerck)	Predatory	14	14	3	33	24	14	102
Thomisidae	*Xysticus kochi*⁺	Thorell	Predatory	8	5	4	2	1	5	25
Zodariidae	*Zodarion italicum*⁺	(Canestrini)	Predatory	1	13	4	0	10	2	30
Zoridae	*Zora spinimana*	(Sundevall)	Predatory	43	2	4	22	8	5	84

Table A9.5.3 Capture data for aculeate Hymenoptera from aggregate mounds experiment at Hadleigh Park, Pitsea Landfill, and Rainham Marshes, in 2015 and 2016. Treatments were 5-tonne mounds of recycled bulk fill (n=9) or Thanet sand (n=9), and unmanaged grassland (n=9). There was zero data from unmanaged grassland plots. This was based on 30 minutes hand netting per plot during three visits each year, in May, June, and August. Female aculeates excavating or provisioning nest holes, or quartering low over the plot surface, were captured and identified to species. Male aculeates do not form part of the analysis.

			2015		2016		
Family	Taxon	Authority	RBF	Thanet	RBF	Thanet	Total
Andrenidae	*Andrena chrysosceles*	(Kirby)	0	0	0	1	1
Andrenidae	*Andrena dorsata*	(Kirby)	1	0	0	0	1
Andrenidae	*Andrena flavipes*	Panzer	2	0	1	3	6
Andrenidae	*Andrena haemorrhoa*	(Fabricius)	0	1	0	2	3
Andrenidae	*Andrena labialis*	(Kirby)	0	1	0	0	1
Andrenidae	*Andrena minutula*	(Kirby)	3	1	1	8	13
Andrenidae	*Andrena nigroaenea*	(Kirby)	0	1	0	0	1
Andrenidae	*Andrena nitida*	(Muller)	0	0	2	1	3
Andrenidae	*Andrena praecox*	(Scopoli)	0	0	0	1	1
Andrenidae	*Andrena scotica*	Perkins, R.C.L.	0	0	2	1	3
Apidae	*Anthophora bimaculata*	(Panzer)	1	0	0	0	1
Apidae	*Nomada flava*	Panzer	0	0	0	1	1
Apidae	*Nomada flavoguttata*	(Kirby)	1	7	9	7	24
Apidae	*Nomada flavopicta*	(Kirby)	0	1	0	1	2
Apidae	*Nomada goodeniana*	(Kirby)	0	2	1	0	3
Apidae	*Nomada marshamella*	(Kirby)	0	0	2	1	3
Colletidae	*Hylaeus communis*	Nylander	2	0	0	0	2
Colletidae	*Hylaeus hyalinatus*	Smith	2	0	0	0	2
Crabronidae	*Cerceris arenaria*	(Linnaeus)	0	1	0	0	1
Crabronidae	*Cerceris quinquefasciata*	(Rossius)	0	0	0	2	2
Crabronidae	*Cerceris rybyensis*	(Linnaeus)	1	1	0	1	3
Crabronidae	*Crossocerus wesmaeli*	(Vander Linden)	0	1	0	0	1

Family	Taxon	Authority	2015		2016		Total
			RBF	Thanet	RBF	Thanet	
Crabronidae	*Entomognathus brevis*	(Vander Linden)	0	8	0	5	13
Crabronidae	*Gorytes quadrifasciatus*	(Fabricius)	0	2	0	0	2
Crabronidae	*Mimumesa unicolor*	(Vander Linden)	0	3	0	0	3
Crabronidae	*Oxybelus uniglumis*	(Linnaeus)	2	36	3	11	52
Crabronidae	*Philanthus triangulum*	(Fabricius)	0	1	0	0	1
Crabronidae	*Trypoxylon medium*	de Beaumont	1	3	0	0	4
Halictidae	*Halictus rubicundus*	(Christ)	0	3	0	5	8
Halictidae	*Lasioglossum albipes*	(Fabricius)	0	1	1	0	2
Halictidae	*Lasioglossum calceatum*	(Scopoli)	1	0	1	1	3
Halictidae	*Lasioglossum leucozonium*	(Schrank)	0	1	0	4	5
Halictidae	*Lasioglossum malachurum*	(Kirby)	2	2	2	1	7
Halictidae	*Lasioglossum minutissimum*	(Kirby)	5	29	4	22	60
Halictidae	*Lasioglossum morio*	(Fabricius)	4	0	5	1	10
Halictidae	*Lasioglossum parvulum*	(Schenck)	0	2	2	0	4
Halictidae	*Lasioglossum pauperatum*	(Brulle)	2	4	3	6	15
Halictidae	*Lasioglossum pauxillum*	(Schenck)	3	2	4	2	11
Halictidae	*Lasioglossum puncticolle*	(Morawitz)	8	0	3	2	13
Halictidae	*Lasioglossum villosulum*	(Kirby)	7	5	0	3	15
Halictidae	*Sphecodes ephippius*	(Linnaeus)	0	4	0	1	5
Halictidae	*Sphecodes geoffrellus*	(Kirby)	2	0	0	3	5
Halictidae	*Sphecodes gibbus*	(Linnaeus)	0	1	0	0	1
Halictidae	*Sphecodes longulus*	von Hagens	0	1	0	0	1
Halictidae	*Sphecodes monilicornis*	(Kirby)	0	1	0	0	1
Halictidae	*Sphecodes puncticeps*	Thomson	0	6	0	1	7
Melittidae	*Melitta leporina*	(Panzer)	0	1	0	0	1
Pompilidae	*Anoplius concinnus*	(Dahlbom)	0	0	1	0	1
Pompilidae	*Arachnospila anceps*	(Wesmael)	0	4	0	1	5

Family	Taxon	Authority	2015		2016		Total
			RBF	Thanet	RBF	Thanet	
Pompilidae	*Auplopus carbonarius*	(Scopoli)	1	0	0	0	1
Pompilidae	*Episyron rufipes*	(Linnaeus)	0	6	0	0	6
Pompilidae	*Priocnemis pusilla*	Schioedte	0	4	0	0	4
Sphecidae	*Ammophila sabulosa*	(Linnaeus)	0	4	0	0	4